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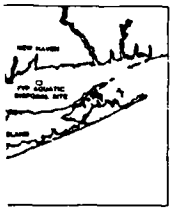

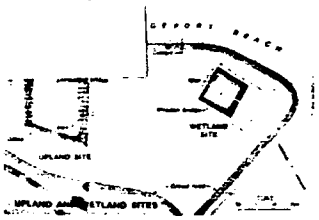

FIELD VERIFICATION PROGRAM
(WETLAND DISPOSAL)

TECHNICAL REPORT D-89-2

SYNTHESIS OF THE RESULTS OF THE FIELD VERIFICATION PROGRAM WETLAND DISPOSAL ALTERNATIVE

by

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19. ABSTRACT (Continue on reverse if necessary and identify by block number)

Contaminated dredged material from the Black Rock Harbor (BRH) maintenance dredging project of the US Army Engineer Division, New England, was used to field verify: (a) procedures for predicting contaminant mobility into plants and (b) procedures for predicting contaminant mobility into animals.

The wetland disposal site was constructed within a protected area using conventional construction techniques and was hydraulically filled from barges. The filling operation provided conditions typical of confined dredged material disposal operations. Following filling, a weir allowed free drainage of surface water (as with most disposal sites) as the fill stabilized through consolidation. Within approximately 9 months, the substrate had stabilized at the desired surface elevation.

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19. ABSTRACT (Continued).

The estuarine plant bioassay procedure was used in the laboratory to evaluate heavy metal uptake by plants from composited BRH sediment. The chemical and laboratory portions of the bioassay were used to predict potential plant uptake of heavy metals. In general, the laboratory plant bioassay adequately predicted heavy metal content of field-grown *Spartina alterniflora* during the first 2 years following establishment of the wetland.

The wetland animal bioassay procedure was applied to composited BRH sediment in the laboratory using an adaptation of the plant bioassay apparatus. This adaptation was unsuitable because of high sandworm mortality during the test. Another study conducted with fresh sediment indicated that the BRH sediment had to be diluted for the animals to survive the static conditions of the test. A preliminary dynamic, tidal simulation rather than static bioassay was conducted using undiluted BRH sediment. The wetland animals not only survived in the simulated tidal bioassay, but results indicated that there was a potential for accumulation of heavy metals and organic contaminants.

A second series of animal bioassays using tidal simulation was used with dredged material collected from the field site. Results of this series of bioassays agreed with the first series of simulated tidal bioassays; there is a potential for wetland animals in contact with contaminated dredged material to accumulate contaminants.

Comparison of field-collected animal data with the laboratory tidal bioassay data suggests that tidal simulation bioassay procedures are somewhat overpredictive of organic contaminant bioaccumulation. No clear pattern between laboratory data and either of the field experiments emerged for heavy metals. Additional testing is recommended before an appropriate wetland animal bioassay procedure can be recommended.

EXECUTIVE SUMMARY

The US Army Corps of Engineers (CE) uses bioassay results and other factors to determine the disposal alternative with the least potential for contaminant mobility. Some of these evaluations are made using first-generation techniques that have not been field verified and therefore generate data whose interpretation is subject to disagreement. For this reason, the CE/US Environmental Protection Agency Interagency Field Verification of Testing and Predictive Methodologies for Dredged Material Disposal Alternatives Program (Field Verification Program or FVP) was established.

As part of the FVP, contaminated sediment was collected from Black Rock Harbor (BRH), near Bridgeport, Conn., prior to scheduled maintenance dredging. The sediment was transported to the US Army Engineer Waterways Experiment Station where it was mixed and subdivided among the participating research groups. Physical, chemical, and biological testing procedures were applied to predict the results of placement of the dredged material in aquatic, upland, and wetland creation disposal environments. This report presents the results of laboratory and field studies of wetland plant and animal bioassay procedures and compares the laboratory results to those obtained in field studies.

The estuarine plant bioassay procedure was used in the laboratory to evaluate contaminant uptake by plants from composited and homogenized BRH sediment. Chemical analysis, including chemical extraction procedures, of the freshly composited BRH sediment was conducted as part of the plant bioassay procedure to predict potential plant uptake of heavy metals. In general, the laboratory plant bioassay adequately predicted heavy metal content of field-grown *Spartina alterniflora* during the first 2 years following establishment of the wetland. It may be desirable to continue field verification of laboratory results for at least a 5-year period to ensure that predictions are accurate once the created wetland has stabilized physically and geochemically. However, heavy metal content of field-grown plants from the FVP wetland was not significantly higher than that of plants in natural marshes from surrounding areas or growing at the site prior to FVP wetland construction, with the possible exception of chromium. These results indicate that the short-term assessment conducted in this study may be sufficient for verification of laboratory predictions for BRH sediment. Previous research indicated that

S. alterniflora does not accumulate organic contaminants; therefore, analyses for polychlorinated biphenyls (PCBs) or polynuclear aromatic hydrocarbons (PAHs) were not pursued.

A wetland animal bioassay was also applied to the composited sediment, using an adaptation of the apparatus employed in the plant bioassay procedure and an index animal, the sandworm (*Nereis virens*). However, this adaptation was unsuitable because of high sandworm mortality during the test. A substrate modification study conducted with the fresh sediment in the laboratory indicated that a mixture of 25 percent BRH sediment and 75 percent clean sand would permit up to 14-day survival of the bioassay animals under static test conditions. Chemical analysis indicated that cadmium and copper were accumulated in animals exposed to the modified BRH sediment composite. Organic contaminants (PCBs or PAHs) were not accumulated above detection limits.

Subsequent to the initial laboratory static bioassays, the first of two tidal simulation bioassays using undiluted sediment was conducted. This preliminary study indicated that under intertidal conditions there was no toxicity to sandworms or to mud snails (*Nassarius obsoletus*), but bioaccumulation of metals, PAHs, and PCBs occurred in both species. These test results indicated that the potential for accumulation of heavy metals and organic contaminants by wetland animals existed in wetlands created with BRH dredged material.

Following dredging and disposal activities, dredged material collected from the field site was used in a second laboratory bioassay testing procedure using tidal simulation. Use of site material in laboratory testing deviated from the underlying premise of the FVP, i.e., to make predictions derived from tests on sediment collected prior to dredging. However, prolonged development of laboratory wetland animal bioassay procedures and depletion of sediment collected for laboratory analysis precluded this option. Results of this second series of animal tidal bioassays suggested, as did the first tidal bioassay procedure, that there is a potential for wetland animals in contact with contaminated dredged material to accumulate contaminants.

Comparison of field-collected animal data with laboratory tidal bioassay data suggests that tidal simulation bioassay procedures are somewhat overpredictive of organic contaminant bioaccumulation. No clear pattern between laboratory data and either of the field experiments emerged for metals.

Tidal simulation (a modified static renewal bioassay) was a superior test compared to static bioassay procedures with no tidal exchange. Additional testing is required before an appropriate wetland bioassay procedure can be recommended. However, it may be possible to extrapolate results from existing flow-through aquatic bioassays rather than also conducting tidal simulation tests to predict contaminant uptake by biota in the intertidal wetland zone. Tidal simulation and flow-through bioassays should be compared with each other and with field results to determine if such extrapolation is possible for certain species such as sandworms or mussels.

PREFACE

This study was conducted by the Environmental Laboratory (EL) of the US Army Engineer Waterways Experiment Station (WES), Vicksburg, Miss., during the period 1982 to 1986. The study was part of the Interagency Field Verification of Testing and Predictive Methodologies for Dredged Material Disposal Alternatives Program (Field Verification Program or FVP). The program was sponsored by the Headquarters, US Army Corps of Engineers (HQUSACE), and was assigned to the WES under the auspices of the Environmental Effects of Dredging Programs (EEDP). The HQUSACE Technical Monitors for FVP were Drs. William L. Klesch and Robert J. Pierce and Mr. David B. Mathis. The aquatic portion of the FVP study was conducted by the US Environmental Protection Agency Environmental Research Laboratory, Narragansett, R. I.; the wetland and upland portions were conducted by the WES.

The study was conducted by personnel of the Plant Bioassay Team, the Ecosystem Biomonitoring Team, the Surface Runoff and Restoration Team, and the Data Management Team of the Contaminant Mobility and Regulatory Criteria Group (CMRCG), Ecosystem Research and Simulation Division (ERSD), and by the Water Resources Engineering Group and the Water Supply and Waste Treatment Group of the Environmental Engineering Division (EED), WES. The Plant Bioassay Team was composed of Dr. Bobby L. Folsom, Jr., Team Leader, the late Ms. Karen M. Garner (Preston), Ms. Cynthia L. Teeter, and Ms. Joycie R. Bright; Dr. Judith C. Pennington and CAPT Todd R. Higgins joined the team toward the end of the project. The Ecosystem Biomonitoring Team was composed of Dr. John W. Simmers, Team Leader, Ms. Carole P. Brown, Mr. Peter J. Pikul, Mr. J. Morris Richards, Mr. R. Glenn Rhett, Ms. Susan A. Portzer, and Ms. Mary Anne Tweedle; Dr. Henry E. Tatem joined the team in the latter phase of the program. Statistical analysis was provided by Mr. Dennis L. Brandon and Ms. Joan U. Clarke of the Data Management Team. Dr. James M. Brannon and Ms. Clarke provided technical review and comment.

The report was written by Dr. Simmers, Mr. Rhett, Dr. Stratford H. Kay, and Dr. Folsom, and was edited by Ms. Jessica S. Ruff of the WES Information Technology Laboratory.

The study was conducted under the general supervision of Dr. Charles R. Lee, Chief, CMRCG; Mr. Donald L. Robey, Chief, ERSD; Dr. Raymond Montgomery, Chief, EED; and Dr. John Harrison, Chief, EL. Dr. Robert M. Engler was EEDP

Manager at the completion of the study; Mr. Charles C. Calhoun, Jr., was the previous Program Manager. The FVP Coordinator was Mr. Robert L. Lazor. Dr. Thomas D. Wright was the Technical Coordinator for the FVP reports.

Appreciation is expressed to Ms. Martha R. Barton, Mr. Horace C. Allen, Mr. James R. Kemp, and Mr. Gary Emerson of the Plant Bioassay Team for their help in conducting some of the experimentation during the course of this study. The assistance of Dr. J. M. Marquenie of the Technology for Society Division (MT) of the Netherlands Organization for Applied Scientific Research is greatly appreciated. Appreciation is also expressed to Dr. Lance L. Stewart, University of Connecticut, for assistance in establishing the wetland creation, and to Messrs. Martin Brodie, Walter Rayford, and Donald K. Crawley, ERSD, for assistance in the laboratory and field studies. The authors offer their appreciation to United Illuminating Company for assistance and support of onsite activities. Technical advice was received from the Ecosystem Bio-monitoring Team's Terrestrial Animal Bioassay Working Group.

COL Dwayne G. Lee, EN, was Commander and Director of WES. Dr. Robert W. Whalin was Technical Director.

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CONTENTS

SYNTHESIS OF THE RESULTS OF THE FIELD VERIFICATION PROGRAM WETLAND DISPOSAL ALTERNATIVE

PART I: INTRODUCTION

Background

1. The Long-Term Effects of Dredging Operations Program, and similar research preceding it under the Dredging Operations Technical Support Program criteria funding, has conducted and will continue to conduct the essential first steps of research and development of theoretically sound and practical evaluative techniques. For those techniques that have been developed, it is essential for their acceptance by other regulatory and resource agencies to document and verify under field conditions both the accuracy of the techniques and the overall environmental consequences of the predicted changes. To meet these needs, the Field Verification Program (FVP), a cooperative effort between the Corps and the US Environmental Protection Agency, was instituted to provide the field with verified procedures and interpretative guidance for use in assessing the environmental consequences of dredged material disposal under aquatic, wetland, and upland conditions. The US Army Engineer Waterways Experiment Station (WES) was the lead Corps laboratory and was responsible for the wetland and upland portions of the program. This report examines only the wetland portion.

2. When creating a wetland with dredged material, a major concern that must be addressed by predictive tests is the potential movement of contaminants into the environment through plants and animals inhabiting the resulting wetland. Experience at the WES has shown that these concerns can best be addressed by bioassay procedures. A bioassay is a test procedure in which organisms or plants are exposed to the test material under controlled laboratory conditions and the resulting toxicity and contaminant uptake are assessed.

3. The wetland animal studies in the FVP are unique in that standard dredged material test procedures did not exist at the initiation of the program for the wetland animal bioassay as they did for the plant bioassay. A modification of the test procedures for wetland plants was thought to be

appropriate for bioassay testing of wetland animals. However, when the initial test procedures proved unacceptable, it became necessary to develop predictive test procedures for wetland animals concurrently with field studies. Therefore, the studies reported here for wetland animal bioassays focus on test development in addition to field assessment of laboratory test results.

Objective

4. The objective of this study was to compare field data from a wetland creation site with results of predictive tests for plant and animal contaminant uptake. Because of the concurrent field testing and initial development of the wetland animal bioassay, information on test modification and initial development for this procedure is also provided.

The FVP Field Site

Construction

5. The location of the FVP field site is shown in Figure 1. Details of the FVP wetland field site are illustrated in Figure 2. The site (hereafter

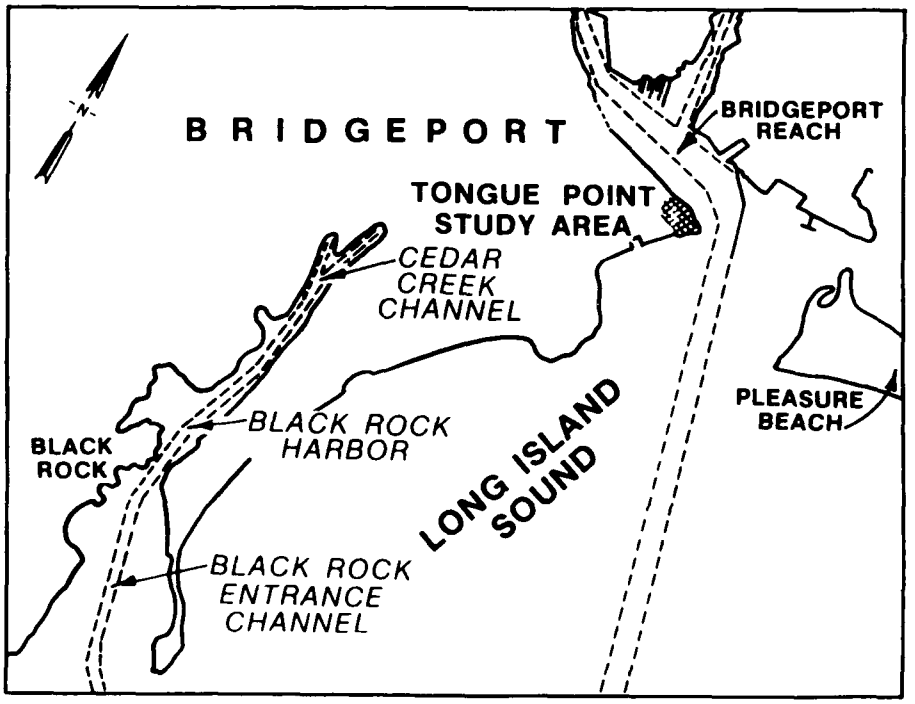


Figure 1. Location of the FVP field site, Bridgeport, Conn.

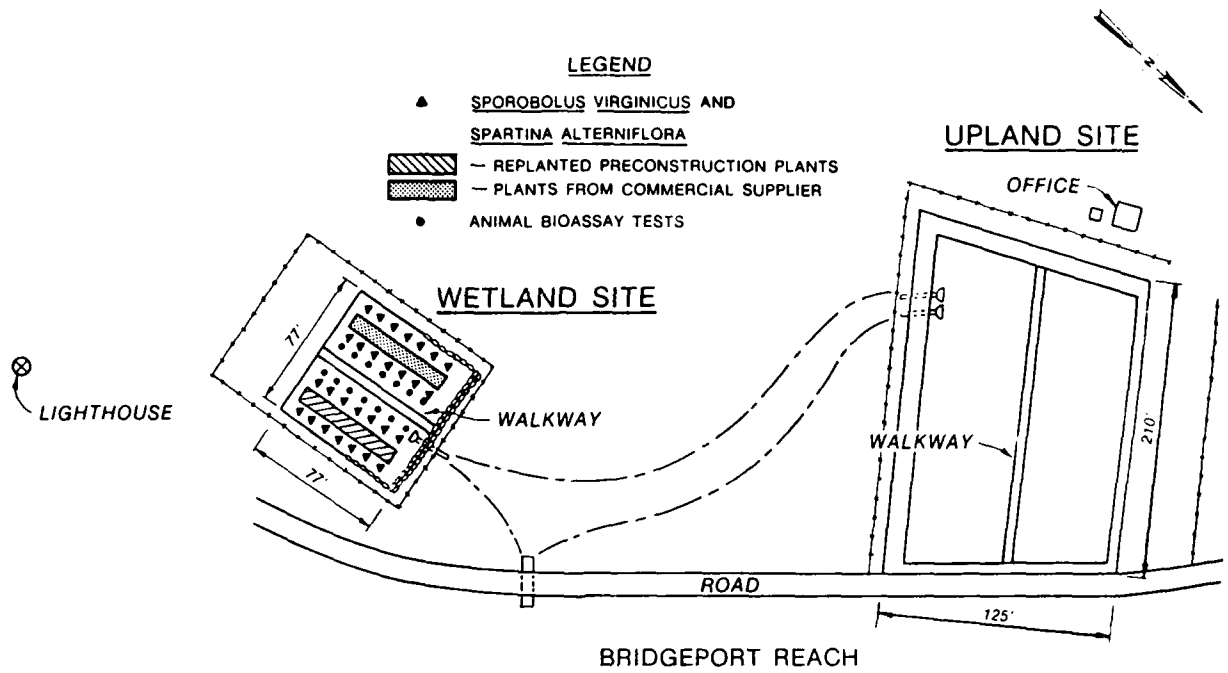


Figure 2. Schematic of the wetland field site (to convert feet to metres, multiply by 0.3048)

referred to as "wetland") is located at Tongue Point adjacent to Bridgeport Harbor, Connecticut, and approximately 4.5 nautical miles (8.3 km) northeast of Black Rock Harbor (BRH), Connecticut. The site is located on the property of United Illuminating Company, a local electrical generating plant, and was leased by the US Army Engineer Division, New England, for the duration of the FVP studies. Prior to construction, the site had been used as a dump and contained some building rubble. The site is surrounded by a dike and road separating it from Bridgeport Reach on Long Island Sound. A culvert beneath the dike allows daily tidal flux. Prior to construction, the area consisted of open water and a surrounding *Spartina alterniflora* marsh grading into an upland stand of *Phragmites australis*.

6. Prior to construction of the wetland, *S. alterniflora* was collected from 650 sq m of the *S. alterniflora* marsh. The *S. alterniflora* sod was removed and transported to the University of Connecticut Marine Science Laboratory greenhouse at Avery Point (near Groton, Conn.). The sod was placed in a greenhouse (with neither auxiliary heat nor lighting) on plastic sheeting and was watered with seawater once weekly for 4 months until replanting.

Following site construction, the plants were transplanted to the newly created wetland.

7. The wetland was created by grading and dike construction with material excavated from the existing intertidal zone. Designs for sedimentation and storage followed recently developed Corps procedures (Palermo, Montgomery, and Poindexter 1978; Palermo 1985). The installation of a weir allowed the daily tidal cycle to flood and drain the wetland. A foot bridge was constructed across the site to allow access without significant disturbance to the dredged material while conducting field bioassays. Dredged material from BRH was then pumped into the site and allowed to consolidate to a depth of approximately 1 m. The filled site occupies approximately 700 sq m. Complete details of site construction and filling are available elsewhere (Folsom et al., in preparation).

Sediment collection

8. The collection and homogenization of the large quantity of sediment from BRH necessary for the initial laboratory predictive tests are described in Folsom et al. (in preparation). A separate barrel of sediment was collected from an area near the mouth of BRH for use as a reference sediment (referred to as Black Rock reference (BRR) sediment) in the animal bioassay tests. It was included in the testing to distinguish the effects of contamination on test results from the physical effects of particle size. Following the selection of the FVP field site at Tongue Point, sediment samples were collected (prior to construction) from the existing intertidal wetland in August 1983 for use in additional animal bioassay tests. In November 1983, immediately after the construction and filling of the wetland with the sediment dredged from BRH, freshly deposited dredged material was collected, placed into sealed containers, and transported to the WES for subsequent tidal simulation animal bioassay tests. The sealed containers of dredged material were refrigerated at approximately 4° C until testing was initiated.

PART II: MATERIALS AND METHODS

Laboratory Procedures

Plant bioassay

9. The estuarine plant bioassay procedure, which includes both bioassay and organic chelate diethylenetriaminepentaacetic acid (DTPA) extraction of the sediment (Folsom and Lee 1985), was used to evaluate contaminant uptake from BRH sediment by plants. Bioassays were conducted in experimental units (Figure 3) similar to those used by Folsom and Lee (1981a). Each of the experimental units was filled with the composited BRH sediment and planted with five germinated seeds of *S. alterniflora* and five sprigs of *Sporobolus virginicus*. The plants were allowed to grow to maximum vegetative growth

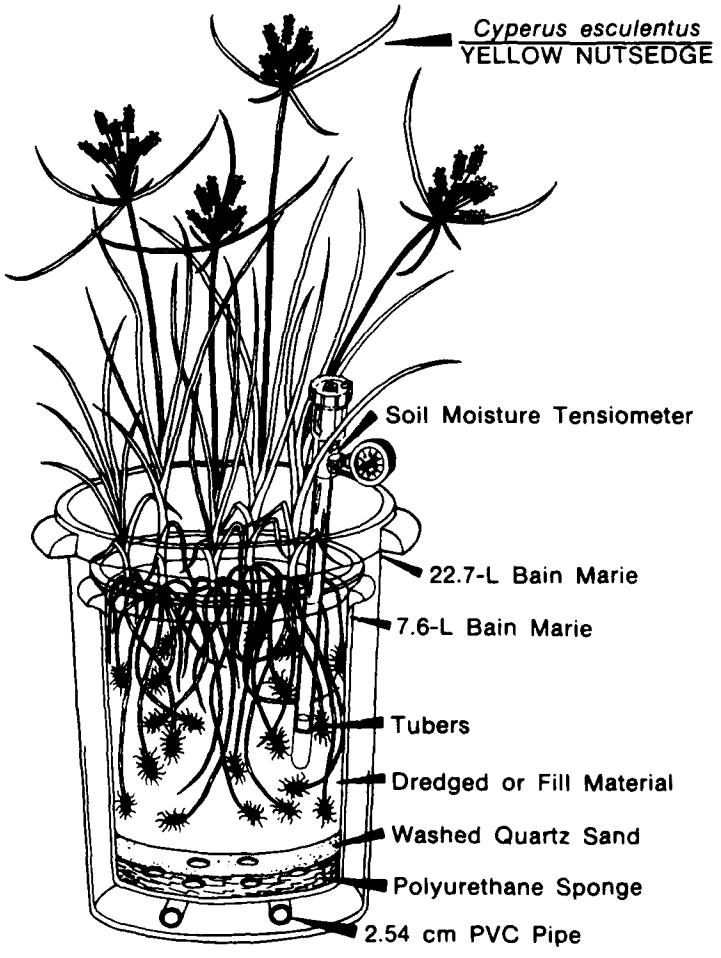


Figure 3. Schematic of the plant bioassay apparatus used in the study

(90 days after planting) at which time they were harvested. The flooded sediments were not allowed to drain or dry out. A 5-cm depth of 15 ppt salinity deionized water-Instant Ocean® saltwater was maintained over the sediment surface by the addition of the 15 ppt saltwater solution as necessary.

10. Harvested plant material was dried to constant weight at 70° C. The harvested plant material was digested in nitric acid-red fuming nitric acid, and subsequently analyzed for zinc, cadmium, copper, nickel, chromium, and lead. All concentrations are reported on an oven-dry weight basis.

11. The BRH sediment was analyzed for texture, organic matter, electrical conductivity, calcium carbonate equivalent, lime requirement, pH, total sulfur, oil and grease, and heavy metals (total nitric acid digestible and DTPA extractable). The total nitric acid digestion provides an estimate of the total amount of heavy metals present in the sediment. Heavy metal extraction by DTPA provides an estimate of that fraction of the total heavy metal content of a sediment that is available to plants (Folsom and Lee 1981a, 1981b; Folsom, Lee, and Bates 1981). The sediment data were used in the equations of Lee, Folsom, and Bates (1983) for predicting heavy metal uptake by plants.

Animal bioassay organisms

12. The sandworm (*Nereis virens*) was selected as a "standard" index organism for the initial bioassay test. The organism is easily obtainable from commercial bait suppliers, provides sufficient biomass for chemical analysis, and has been used frequently in aquatic bioassay procedures (McLeese, Metcalfe, and Pezzack 1980; Rubinstein, Lores, and Gregory 1983). Following selection of the wetland and completion of background studies, the mud snail (*Nassarius obsoletus*) and the ribbed mussel (*Modiolus demissus*) were added as additional indicator species (potential index species). The mud snail had not been used previously as a bioassay organism but was selected because of its abundance at the FVP site and on intertidal mud flats in surrounding areas. The mud snail generally remains submerged and often burrows within the surface sediments. It is one of the dominant invertebrate species found in the Connecticut salt marshes and feeds at the sediment surface (Brousseau 1981, Fell et al. 1982). Ribbed mussels were selected to represent the nonburrowing, filter-feeding component because of their great abundance on intertidal rocky areas along the coastline of New England. These organisms are also important

and abundant in salt marshes (Fell et al. 1982) and occur naturally both within and adjacent to the FVP site.

13. Sandworms used for the laboratory bioassays were supplied by Mr. Ivan Fly of New Castle, Maine, a commercial bait dealer. The mud snails and mussels were initially field collected near the FVP site. Later efforts were made to collect them from other areas of lesser contamination. All test species were held in the laboratory in tanks containing washed sand and artificial seawater (Carolina Instant Ocean®) at 20° C and 22 ppt salinity. The seawater was changed daily. The sandworms and snails were fed Tetramin tropical fish food, and the mussels were fed Carolina Invertebrate Diet.

Static laboratory bioassays

14. The initial laboratory predictive test for the animal bioassay was a static test procedure similar to the plant bioassay procedure (Folsom and Lee 1981a; Simmers, Rhett, and Lee 1984). Sandworms were exposed to 3.2 ℓ of sediment overlain with 15 cm of continuously aerated artificial seawater (Carolina Instant Ocean®) at 22 ppt salinity in 76-ℓ aquaria. The aquaria were placed in a water bath and maintained at 17° C for the duration of the test. Bioassay units contained either BRH sediment, BRR, or washed sand. Each treatment was replicated four times. Each test unit was initially stocked with 12 individual sandworms, which were subsequently exposed to the test material for 14 days.

15. Although the static procedure initially used for the wetland bioassays had not been as extensively used as comparable aquatic techniques, the 100-percent sandworm mortality in BRH sediment was unexpected. This mortality led to the modification of the BRH sediment bioassay through the addition of various proportions of sand to reduce the concentration of chemical contaminants and/or provide a more suitable physical substrate for the worms. These modified bioassays contained BRH sediment, BRR, or washed sand. Static bioassays and experimental procedures similar to those described in the preceding paragraph were used with the following modifications. Black Rock Harbor sediment was modified with clean washed sand in the following percentages: 0 percent sand, 100 percent BRH; 25 percent sand, 75 percent BRH; 50 percent sand, 50 percent BRH; 75 percent sand, 25 percent BRH; and 100 sand, 0 percent BRH. Sandworm survival after 14 days of exposure to the various mixtures of sand and BRH was observed only in 75-percent sand/25-percent BRH and in 100-percent sand.

16. Following this testing to determine the amount of sand modification necessary for organisms to survive, the initial static bioassays conducted with BRH sediment were repeated using a mixture of 75 percent sand and 25 percent BRH. Bioassay conditions were the same as previously described, including the bioassays with BRR and washed sand. Each test unit was initially stocked with approximately 25 g of sandworms (7 to 13 animals). Each bioassay treatment was replicated four times. At the end of 14 days of exposure, the sandworms were recovered, counted, and weighed as a group. The sandworms were then placed in artificial seawater overnight to purge the gut contents, killed by freezing, and kept frozen until chemical analysis.

Tidal simulation laboratory bioassays

17. The field site was selected following the static bioassays. However, field observations and recommendations from working group experts regarding the need to simulate tidal action led to the modification of the animal bioassay procedure to incorporate a simulated tidal regime. Therefore, subsequent laboratory bioassays were designed to more closely simulate the daily tidal flux and flushing action observed in the BRH wetland. The preliminary tidal simulation bioassay utilized four large tidal simulation chambers (19 × 41 × 120 cm) containing 20 ℓ of sediment and operated to simulate a 12-hr tidal cycle (Figure 4). Peristaltic pumps were used to gradually inundate the sediment over a 6-hr period and drain the sediment in the subsequent 6 hr. Fresh artificial seawater (Carolina Instant Ocean®) at 22 ppt salinity was used and discarded at the end of each tidal cycle. The test chambers contained BRH, BRR, freshly collected background sediment from the preconstruction wetland site, or washed sand as used in the animal bioassay laboratory maintenance cultures. A temperature of 20° C was maintained in the bioassay units by immersing the test chambers in a water bath. Because of the preliminary nature of this test, treatments were not replicated. The BRH and BRR had been stored 15 months (in airtight, sealed containers at 4° C) by the time this test was conducted.

18. In this tidal simulation bioassay, each tidal chamber was stocked with 348 mud snails and 70 sandworms. During the study the animals were fed Tetramin® tropical fish food. The water was aerated using water-filtered air. Dead animals were removed as discovered. Fifty snails and 10 sandworms from each chamber were collected for analysis after 32 days of exposure. The

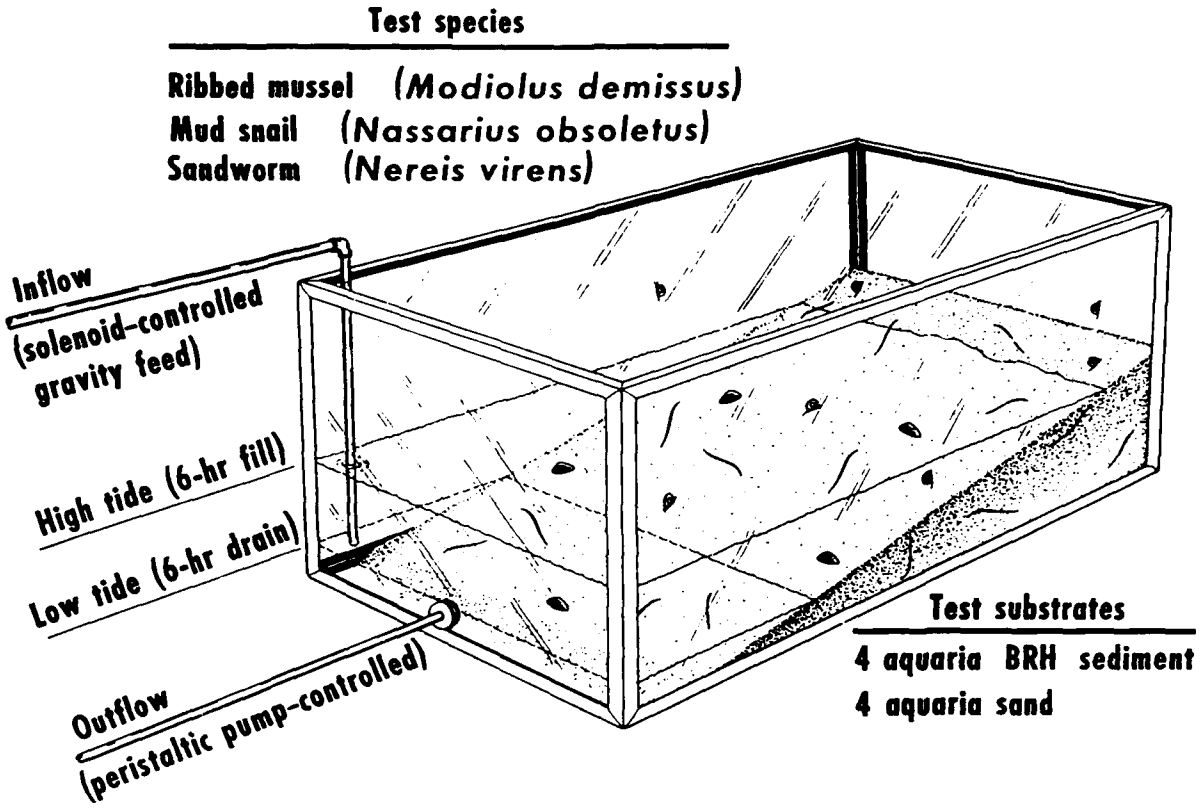


Figure 4. Tidal simulation bioassay apparatus

animals were placed in artificial seawater overnight to purge their gut contents, frozen at -40° C, and stored at -40° C until analysis.

19. A second tidal simulation system was constructed of smaller experimental units so that treatment replication and system flexibility could be increased. Eight 57-ℓ glass aquaria with a gravity flow system for tidal influx and peristaltic pumps for the outflow (Figure 4) were used. Each aquarium contained 5 ℓ of either BRH collected from the wetland or washed sand (as a control). Each treatment was randomly assigned to four replicates. Fifteen sandworms, thirty snails, and five mussels were added to each chamber. Temperature, feeding, and tidal cycle were maintained as previously described. All surviving animals were collected at the end of a 30-day laboratory exposure, purged, and prepared for chemical analysis as previously described. A summary of the four bioassay techniques used is given in Table 1.

Field Procedures

Plants

20. *Spartina alterniflora* sod removed prior to construction was replanted on the freshly deposited dredged material on the east side of the bridge (Figure 2) following construction and filling of the wetland site. *Spartina alterniflora* (supplied by Environmental Concern, St. Michaels, Md.) was planted in the dredged material on the west side of the bridge. *Sporobolus virginicus* (supplied by the University of Delaware) was also planted in the wetland along the west and east sides of the wooden bridge.

21. The aboveground biomass of *S. alterniflora* in the wetland was sampled twice at maximum vegetative growth (September 1985 and September 1986) using the procedures described by Simmers et al. (1981). *Sporobolus virginicus* did not survive past initial planting; therefore, no samples were taken. *Spartina alterniflora* tissue samples were analyzed for the heavy metals zinc, cadmium, copper, nickel, chromium, and lead according to the methods described in the chemical analysis section that follows.

Animals

22. Field animal bioassay studies were initiated after construction and filling of the site. Because survival of animals placed in the field environment was poor due to fish predation, animals were placed in 12-ℓ polyethylene buckets filled to a depth of approximately 10 cm with either BRH material collected from the wetland or a washed sand reference brought from the WES. The buckets were modified to contain screens in the sides and tops. The screens allowed free flow of water across the sediment surface with the daily tidal flux, kept the animals from escaping, and prevented animal predation by fish. The buckets were pushed into the dredged material to the depth of the sediment level inside the bucket. Each bucket contained 10 to 15 sandworms, 30 to 40 snails, or 10 to 20 mussels, depending on availability. Four replicate buckets were used for each species in each of the two treatments. At the end of each test, surviving animals were sorted, counted, and prepared for chemical analysis.

23. The mobility of a second sandworm species (*Nereis succinea*) actively colonizing the wetland was observed using the following method. The wetland site was divided into 1-m grids that provided a point of reference to observe movement of animals. However, collection of animals from the grid was

not done randomly. In addition to observing animal movement, the grid also allowed monitoring of the spread of *S. alterniflora*.

24. The usage of the site by nonresident animals was also observed (Appendix A). Sandworms resident at the site, and present in large numbers, were collected for chemical analysis. The animals were placed into beakers containing Long Island Sound seawater, allowed to depurate for 24 hr, shipped to the WES on ice, and then frozen until prepared for analysis.

Chemical Analyses of Animal Tissues and Substrates

25. All animal tissue samples were homogenized with a Polytron Tissue Homogenizer (Brinkmann Instruments, Inc., Westburg, NY) equipped with a solid titanium milling device prior to analysis. The homogenates were divided into two subsamples, which were used for heavy metal and organic contaminant analysis, respectively. Ash-free dry material and percent water were determined on the homogenates as follows. About 0.5 g of the wet homogenate was weighed to the nearest 0.1 mg into a porcelain crucible and dried to constant weight at 105° C. The oven-dried homogenate was weighed to the nearest 0.1 mg and ashed at 600° C in a muffle furnace. The ash-free homogenate weight was then calculated.

26. Animal tissues, dredged material, and sediments were digested in nitric acid using the procedure of Marquenie, Simmers, and Kay (1987). Digests were analyzed for arsenic, cadmium, copper, chromium, nickel, and lead.

27. Concentrations of polychlorinated biphenyls (PCBs) and hexachlorobenzene were determined in extracts of animal tissues and sediment samples by electron capture gas chromatography, with Mirex as an internal standard, using analytical methods described in detail by Marquenie, Simmers, and Kay (1987). These analyses were conducted on approximately 1 to 4 g of wet animal tissue and 5 g (wet weight basis) of substrate/dredged material. The PCB congeners determined were those selected and recommended by a 1985 Dutch standardization committee on PCB analysis. The selection was based on the frequency of occurrence in contaminated sediments worldwide and the ability of gas chromatography to separate PCB congener peaks.

28. Concentrations of polynuclear aromatic hydrocarbons (PAHs) in animal tissues and substrates were determined by high performance liquid

chromatography, using approximately either 5 g (wet weight) for substrate/dredged material or 2.5 g (wet weight) for tissue, or by gas chromatograph-mass spectrometer (Marquenie, Simmers, and Kay 1987).

Statistical Data Analysis

29. Data from each experiment were analyzed, where appropriate, using one-way analysis of variance (ANOVA) to detect statistically significant differences among treatment means. Prior to ANOVA, the data were tested for violations of the ANOVA assumption of homogeneity of variances, using Hartley's F max test (Winer 1971). If the variances were found to be nonhomogeneous at a significance level of 0.05, the data were transformed and then retested for homogeneity of variance. With these data sets, log 10 or square root transformations were found to be the most successful transformations for correcting variance nonhomogeneity.

30. The ANOVAs on untransformed or appropriately transformed data were performed using the SAS ANOVA procedure (SAS Institute, Inc. 1985). Means for more than two groups were compared using the Waller-Duncan k-ratio t-test (SAS Institute, Inc. 1985), with a k-ratio of 100, to produce statistical groupings of means (indicated by capital letters in the summary tables). Because the k-ratio does not correspond directly to a preset significance level (alpha), the significance of the differences among means may be determined by looking at the probability (P) associated with the ANOVA F statistic. A P value greater than 0.05 is customarily considered to be nonsignificant, although higher or lower significance levels are justified in many situations.

PART III: RESULTS AND DISCUSSION

Comparison Between Laboratory and Field Plant Data

31. Selected physical and chemical parameters of the BRH sediment are listed in Table 2. Concentrations of parameters listed in Table 2 are typical of those previously reported for contaminated sediments (Folsom, Lee, and Bates 1981). Total acid-digestible and DTPA-extractable heavy metal concentrations are listed in Table 3. The total acid-digestible sediment content of heavy metals (Table 3) was typical of contaminated saltwater sediments, except for unusually high levels of copper (Folsom, Lee, and Bates 1981). All of the DTPA-extractable metal concentrations in Table 3 were significantly higher in the air-dried sediment than in the wet sediment, indicating a significant release of available metals upon air-drying of the sediment.

32. Heavy metal content of *S. alterniflora* grown in the wetland sediment in the laboratory and in the field is presented in Table 4 along with concentrations predicted using the DTPA data in the equations of Lee, Folsom, and Bates (1983). Analyses of variance were conducted on the data of Table 4 following a square root transformation to achieve homogeneity of variances. Zinc content of the laboratory-grown plants was not significantly different from that of the 1985 field-grown plants, but was significantly lower than that of the 1986 field-grown plants. Cadmium content of laboratory-grown plants was significantly greater than that of both 1985 and 1986 field-grown plants. Copper and nickel concentrations in laboratory-grown plants were not significantly different from concentrations in field-grown plants. Chromium and lead contents of laboratory-grown plants were lower than those of 1985 field-grown plants; however, these differences were only marginally significant, as evidenced by the F-statistics and their associated probabilities (F and P in Table 4).

33. In comparing plant heavy metal content predicted by the DTPA extraction data with the field data (Table 4), it can be seen that zinc and cadmium levels were significantly overpredictive, whereas copper, nickel, and lead levels were more closely predictive of levels in field-grown plants. Chromium was somewhat underpredicted by the DTPA extraction data.

34. Although not a specific objective of FVP, heavy metal contents of field-grown plants can be related to metal contents of *S. alterniflora* growing

at the FVP site prior to construction of the wetland, or growing in other natural wetlands (Table 5). This comparison demonstrates that heavy metal content of field-grown plants from the FVP wetland site was not significantly higher than that of plants in natural marshes from surrounding areas or growing at the site prior to FVP wetland construction. A possible exception is chromium, which is higher in plants from the wetland in 1985 and 1986 than in plants growing at the FVP site prior to construction, although the significance of this difference in marginal ($P = 0.08$).

35. *Sporobolus virginicus*, the second wetland plant species evaluated in the FVP, grew in the laboratory but did not survive in the wetland. Thus, heavy metal concentrations in the tissues of this plant could be estimated by substituting the DTPA and laboratory data into the equations for *S. alterniflora*, but they could not be field verified. The plant normally grows in higher elevation marsh areas, inland of the intertidal wetland area, and is fairly tolerant of high levels of soil salinity. Tolerance of excessive soil salinity was the reason *S. virginicus* was chosen as a potential bioassay index plant. However, *S. virginicus* may not have been able to tolerate and survive the excessively low sediment redox potential that probably existed in the wetland sediment. *Sporobolus virginicus* may not be the best choice for an additional wetland index plant species, but it did show potential for use as an upland plant index species (Folsom et al. 1988). The *S. virginicus* laboratory data are presented in Table 6.

36. Previous research conducted at the WES laboratory (Folsom 1982, Folsom and Preston 1983) indicated that *S. alterniflora* did not take up PCBs or PAHs; therefore, the plants were not analyzed for PCBs or PAHs. Recent studies offer conflicting evidence as to whether PCBs are taken up by plants. Meredith and Hites (1978) found that trees growing near a PCB-contaminated landfill contained PCBs in their bark, but not in wood rings, which indicates the PCBs were deposited from the atmosphere, not taken up systemically. Buckley (1982) reported that the level of PCBs found in the foliage is mainly due to vapor transport from the soil, rather than to translocation through the plant. Bacci and Gaggi (1985) found that PCB levels of tomatoes grown in PCB-contaminated sand did not correlate with levels of PCBs in the sand and concluded that translocation of PCBs through plant tissues was slight. However, Bush et al. (1986) showed that *Lythrum salicaria* (purple loosestrife) took up 42 PCB congeners from soil. They determined that PCB uptake from ambient air

occurred as well, but the dominant route of uptake was through the plant roots. Earlier literature (Iwata and Gunther 1976, Weber and Mrozek 1979) also indicated that PCBs stopped at the root-peel level and were not translocated into the plant.

37. Plant uptake of PAHs has generally been focused on benzo(a)pyrene because of the carcinogenic potential in humans. The most likely pathways for food contamination by PAHs are direct deposition of the chemicals from the atmosphere onto vegetation and deposition on soil, with subsequent uptake by plants via their root systems (Edwards 1986). Shabad and Cohan (1972) showed that spring wheat absorbed small amounts (<0.200 µg/g) of benzo(a)pyrene and that the majority of it was found in the straw. Wagner and Siddiqi (1970) found similar results for plant uptake of benzo(a)pyrene by winter wheat but increased uptake of 3,4-benzofluoranthene, which was biomagnified more than benzo(a)pyrene; the authors concluded that various PAHs may be taken up by different mechanisms. A study by Blum and Swarbrick (1977) on PAH uptake by several vegetable crops indicated only minor uptake of benzo(a)pyrene (<0.003 µg/g) and may have resulted from atmospheric deposition of benzo(a)pyrene.

38. Recent evidence (Edwards 1986) on actual plant uptake of PAHs using ^{14}C-labeled PAHs demonstrated in a nutrient solution experiment that bush bean plants took up anthracene through their roots and translocated it and its metabolites to other plant organs; concentrations of anthracene in plant parts were 1,834 µg/kg in roots, 1.74 µg/kg in stems, and 0.08 µg/kg in leaves. Edwards (1986) also found evidence of increasing proportions of metabolites in stems and leaves, indicating increased translocation to and/or metabolism of anthracene in these tissues. In light of the relatively low level of PCBs and PAHs in the BRH sediment (4.70 and 37.1 µg/g, respectively), in combination with the evidence that they would most likely not be taken up, it seemed appropriate not to analyze the plants for these compounds.

Animal Bioassay Development

39. The results of the preliminary screening tests and the initial laboratory static test with sandworms indicated that the BRH sediment was toxic to the sandworms under the test conditions employed; thus, BRH sediment was modified with sand to permit sandworm survival. Sandworm mortality was

15, 4, and 0 percent respectively for the BRR, modified BRH, and sand. There were no statistically significant differences between the average individual worm weights at Day 0 and Day 14 (Table 7). Sandworms exposed to the modified BRH had significantly higher tissue levels of the metals cadmium and copper after 14 days than worms exposed to either BRR or sand (Table 8). Tissue residues of chromium and nickel were higher in worms exposed to modified BRH than in worms exposed to BRR. Arsenic, lead, and zinc tissue levels were not significantly different among the three treatments. The PCB and PAH concentrations were below detection limits both in the tissues and in the sediments, with the exception of low levels of some PAHs detected in the modified BRH sediment. The short duration of the test (14 days), the changes in physical characteristics of BRH due to modification with sand, and the static water conditions appeared to make this test inappropriate for predicting long-term biological consequences of a wetland created with contaminated dredged material.

40. In the preliminary tidal simulation bioassay, mud snails exposed to the BRH sediment had slightly higher tissue levels of heavy metals than mud snails exposed to the other substrates (Table 9). Sandworms did not show the same pattern of heavy metal accumulation. These relationships are illustrated by cadmium in Figure 5. Snails collected from Tongue Point and maintained in the laboratory in a washed sand culture medium with artificial seawater already contained more than 2,000 µg/g copper prior to the initiation of the test. Even with these high initial levels, there was further uptake of copper from all substrates. The high concentrations of copper in snails in the sand controls and background animal samples suggested a strong affinity for copper. This may be the result of previous exposure to high concentrations of copper, which can result in high levels of copper in organisms (Bryan and Hummerstone 1971). The range of metals in animal tissues in this study generally falls within the ranges summarized by Eisler (1981) for similar species. Mud snails appear to be more sensitive to substrate concentrations than sandworms and provide a better indication of heavy metal mobilization into the environment through animal uptake.

41. Concentrations of PCBs (sum of nine congeners) were higher in snails and sandworms exposed to BRH than in the corresponding animals exposed to the other test media (Table 9) and were also higher in all animals than in the corresponding exposure substrates (Figure 6). Most PAHs were higher in

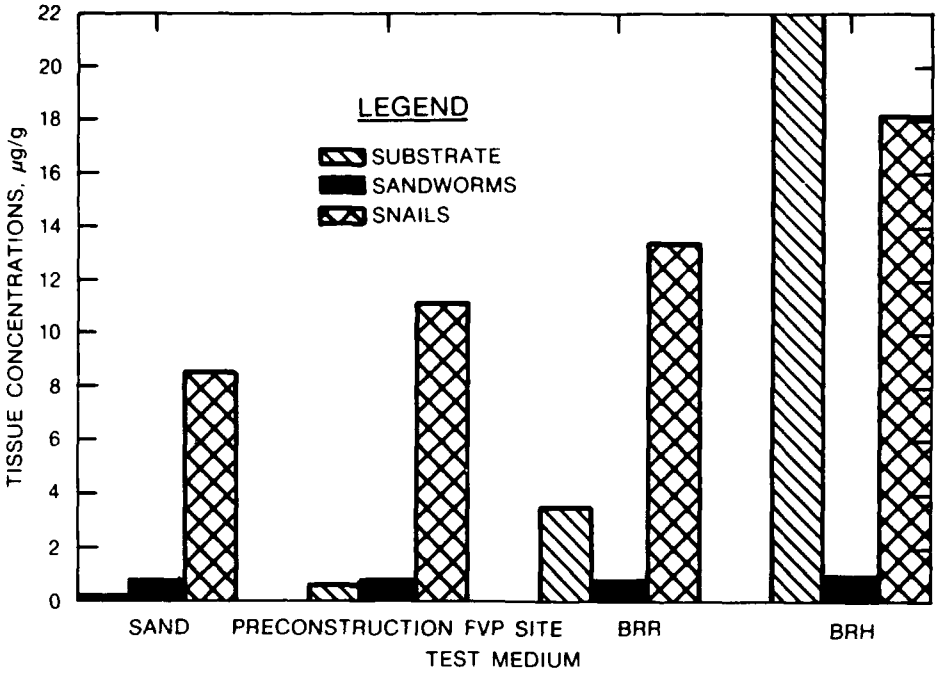


Figure 5. Cadmium in substrates (micrograms per gram dry weight) and animals (micrograms per gram ash-free dry weight) exposed for 32 days in a laboratory tidal simulation chamber

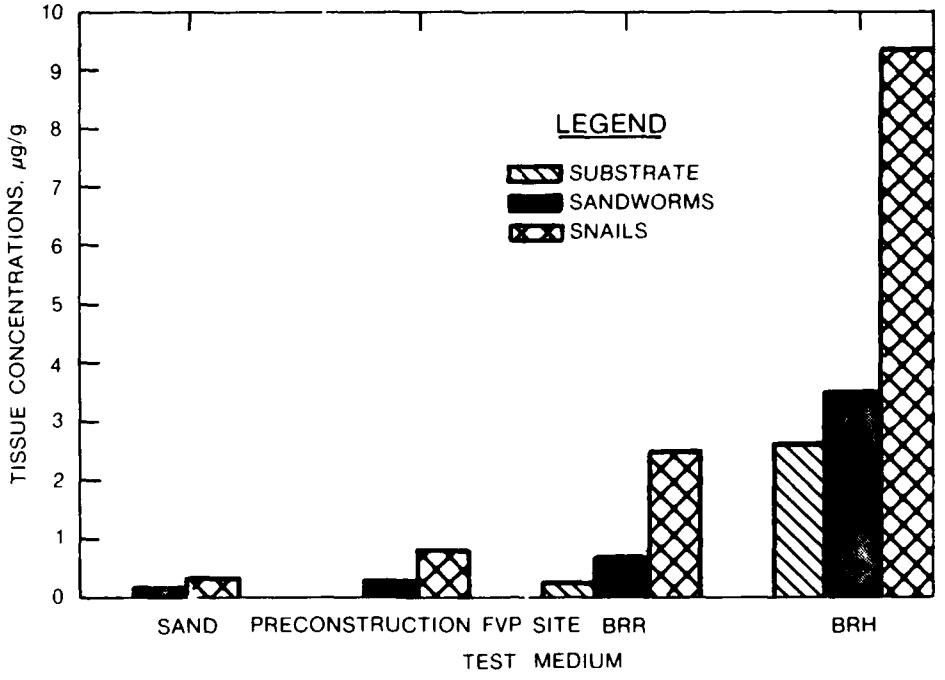


Figure 6. PCBs (sum of nine congeners) in substrates (micrograms per gram dry weight) and animals (micrograms per gram ash-free dry weight) exposed for 32 days in a laboratory tidal simulation chamber

the BRR and BRH substrates than in the animals. The highest tissue levels of most PAHs occurred in animals exposed to BRH. In general, PAH concentrations were higher in snails than in sandworms. These trends are illustrated by phenanthrene in Figure 7. Statistical comparisons could not be made because the treatments (i.e., test media) were not replicated.

42. The results of the preliminary tidal simulation bioassay provide evidence for bioaccumulation of contaminants from BRH. Mud snails appeared to be more sensitive to substrate contaminant concentrations than were sandworms. The data suggest a potential postconstruction route of contaminant mobility into the environment through animal uptake. However, the evidence is inconclusive because test treatments were unreplicated and the substrate had been stored for an extended period. At present, the effects of sediment storage on bioassay results are unknown.

43. Contaminant uptake from the wetland dredged material by sandworms, mussels, and snails was assessed in a second tidal simulation bioassay. In this laboratory test, dredged material collected from the field site was used.

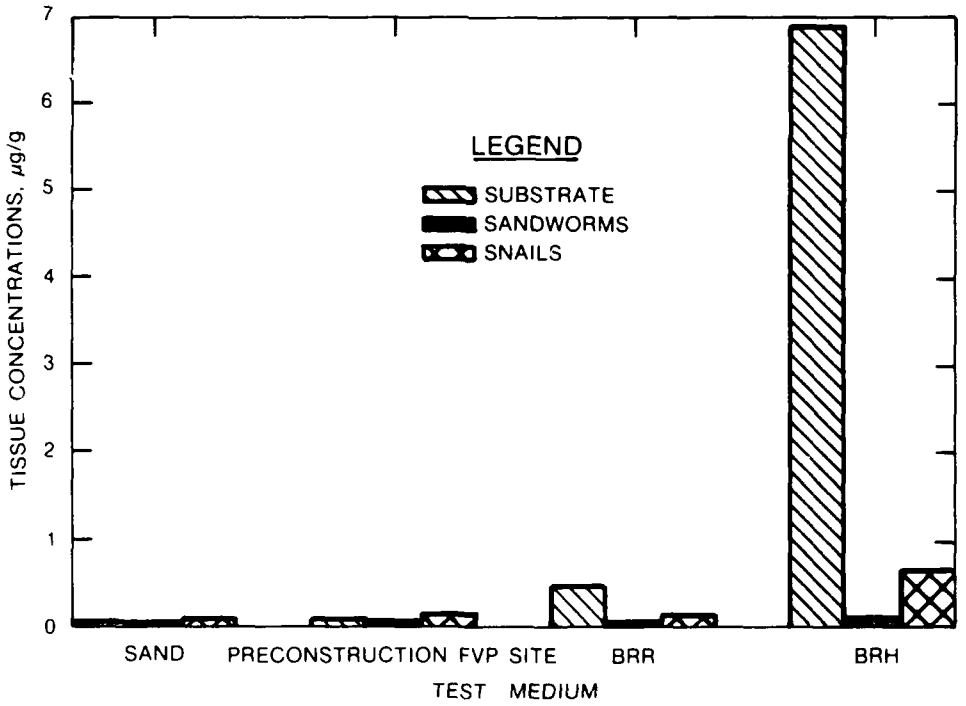


Figure 7. Phenanthrene in substrates (micrograms per gram dry weight) and animals (micrograms per gram ash-free dry weight) exposed for 32 days in a laboratory tidal simulation chamber

Contaminant concentrations in the Black Rock dredged material, and in animals exposed to BRH and to a sand control, are summarized in Table 10 and are presented graphically in Figures 8-12. Statistical comparisons were not made because of the limited replication.

44. The data presented in Figures 8-12 and Table 10 suggest that snails and mussels accumulate metals and organic contaminants to a greater extent than sandworms. Thus, snails and mussels may be better indicator species than sandworms for estimating contaminant uptake by animals. The data also indicate that the sand used as a control contained cadmium as well as some chromium and copper.

45. The data (Tables 9 and 10) from the tidal simulation bioassays suggest that there is a potential for wetland animals in contact with BRH dredged material to take up certain contaminants. These data also underscore the importance of using at least three species in predictive bioassay testing, as

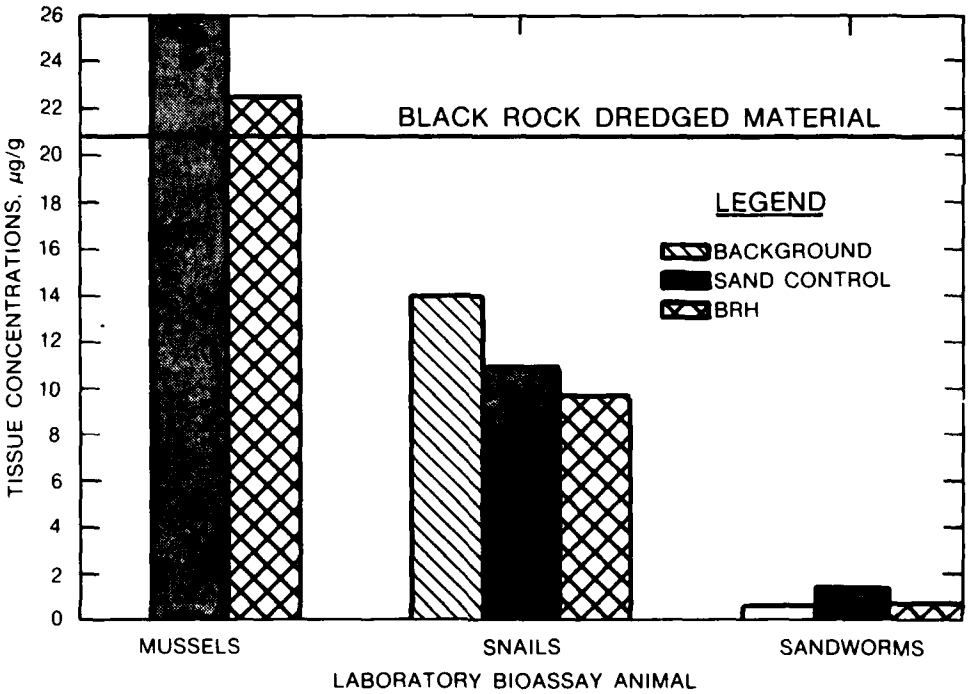


Figure 8. Cadmium in laboratory bioassay animals (micrograms per gram ash-free dry weight) prior to and after 30 days exposure in a laboratory tidal simulation chamber

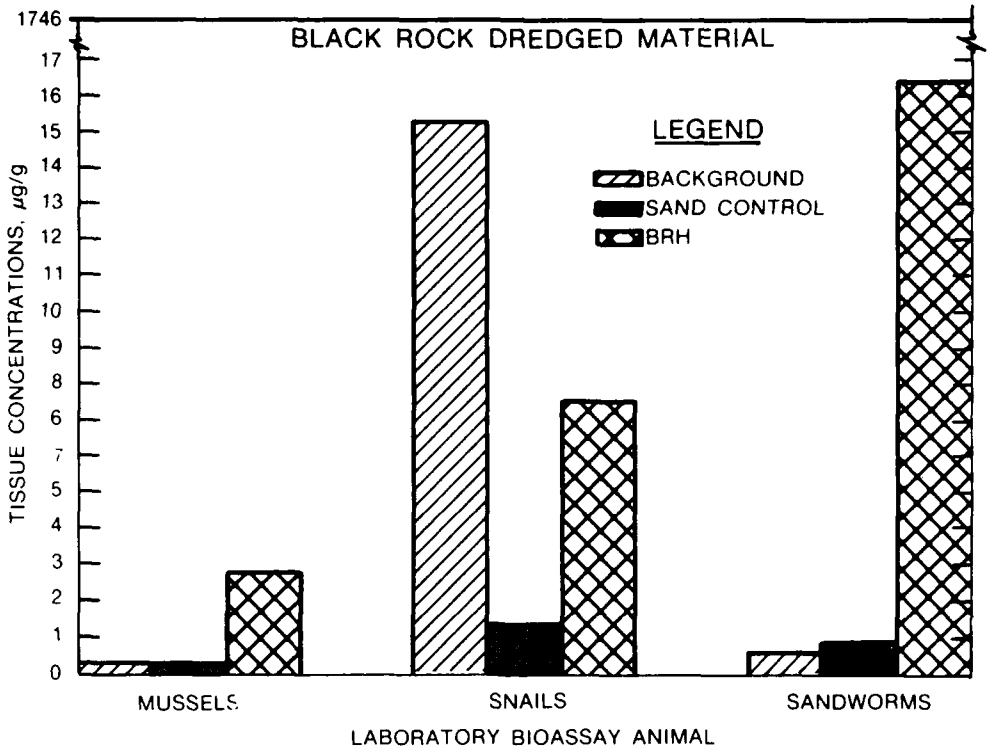


Figure 9. Chromium in laboratory bioassay animals (micrograms per gram ash-free dry weight) prior to and after 30 days exposure in a laboratory tidal simulation chamber

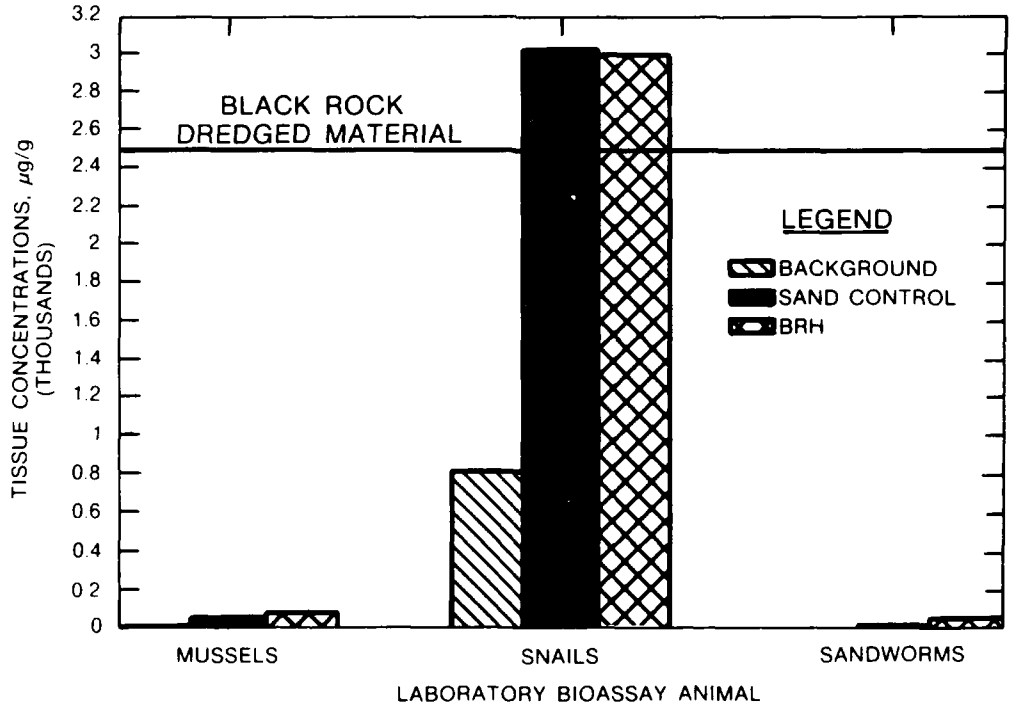


Figure 10. Copper in laboratory bioassay animals (micrograms per gram ash-free dry weight) prior to and after 30 days exposure in a laboratory tidal simulation chamber

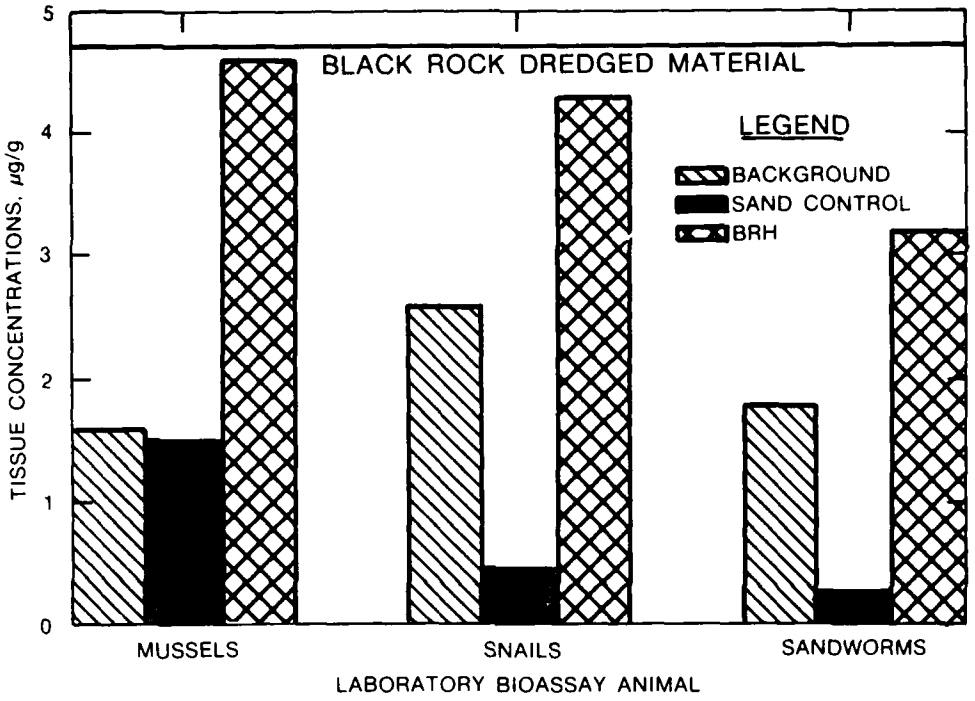


Figure 11. PCBs (sum of 11 congeners) in laboratory bioassay animals (micrograms per gram ash-free dry weight) prior to and after 30 days exposure in a laboratory tidal simulation chamber

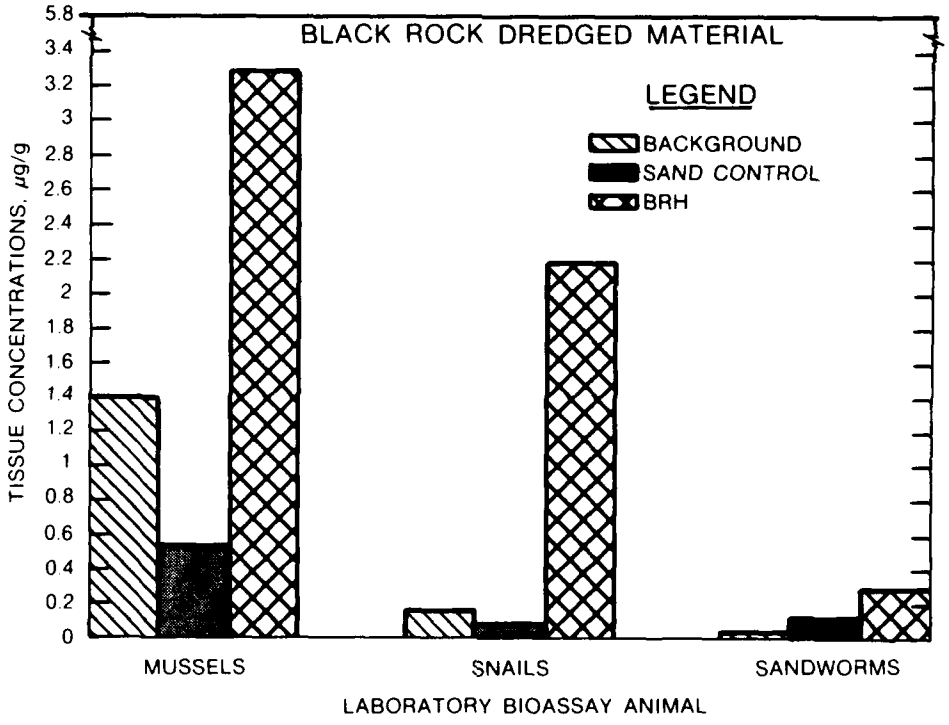


Figure 12. Fluoranthene in laboratory bioassay animals (micrograms per gram ash-free dry weight) prior to and after 30 days exposure in a laboratory tidal simulation chamber

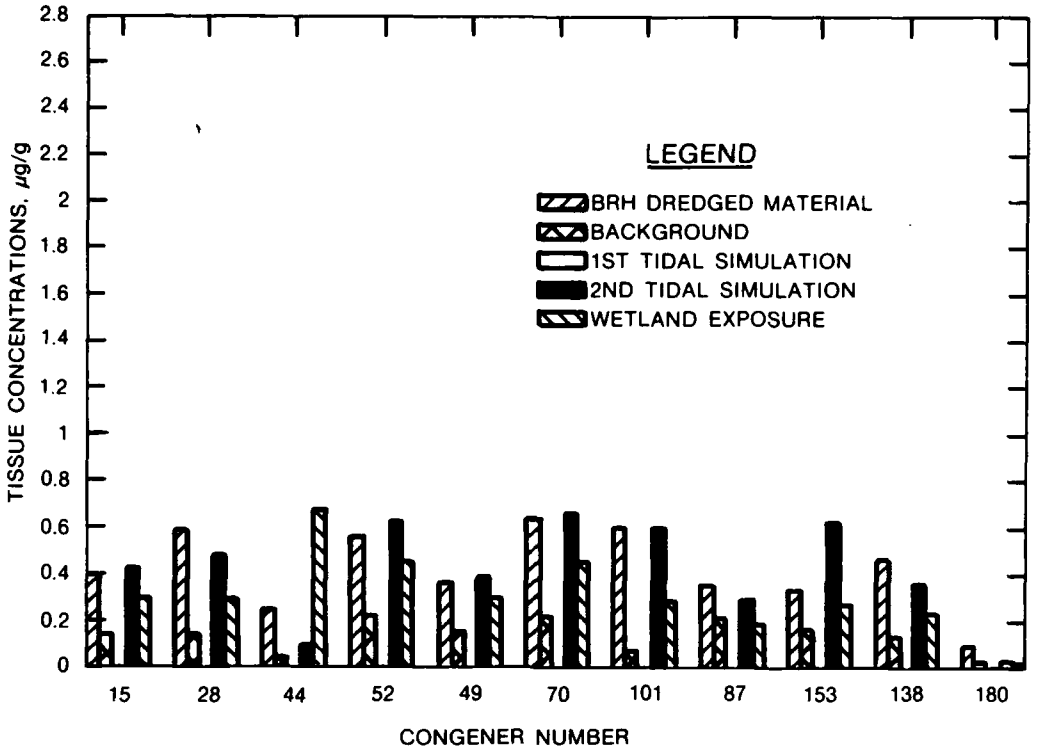
any one or two species may not serve as an indicator of mobility of all contaminants present.

Comparison Between Laboratory and Field Animal Data

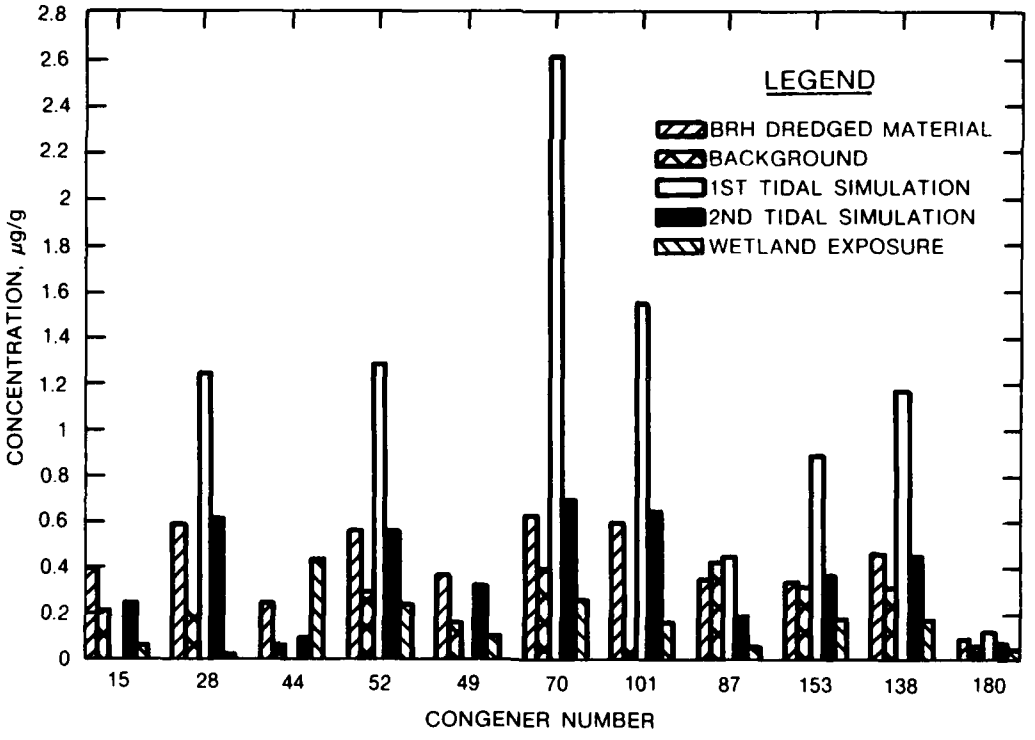
46. Active biomonitoring (emplacing animals at a site and collecting them following exposure) at the field site and at a reference site was not successful. Few full sets of replicates were recovered from the field, regardless of the medium. Analysis of composite samples from the active biomonitoring field tests did indicate, however, that concentrations of most PCB congeners analyzed decreased from background tissue levels in snails over the 28-day field exposure (Table 11). Mussels, however, accumulated 10 of the 11 congeners above background tissue levels, and sandworms accumulated 6 of the 11 congeners above background.

47. A comparison of PCB congener concentrations can be made across experiments using the data presented in Tables 9, 10, and 11. Figure 13 illustrates PCB concentrations in Black Rock dredged material, in background tissue samples, in animals exposed to BRH in the two tidal simulation laboratory experiments (mussels were not included in the preliminary tidal simulation bioassay), and in animals exposed to BRH in the wetland (active biomonitoring). Congener tissue levels in the second tidal simulation experiment tended to approach the corresponding levels in BRH dredged material. Congener tissue levels in the first tidal simulation experiment were almost always greater than those from the second tidal simulation experiment and sometimes (especially in the snails) exceeded the corresponding levels in the substrate. Congener tissue levels from the 28-day field exposure were generally less than corresponding levels in the substrate or from either of the laboratory tidal simulation experiments, with the exception of congener no. 44 (2,3,2',5'-tetrachlorobiphenyl). This suggests that the tidal simulation bioassays are somewhat overpredictive of PCB congener bioaccumulation in animals exposed to BRH in the experimental wetland. The tidal simulation bioassay was likewise overpredictive of hexachlorobenzene and DDE concentrations in field-exposed animals (Tables 10 and 11).

48. Mussel PCB concentrations in the field (Table 11) and laboratory studies (Table 10) were one to two orders of magnitude higher than those reported in the literature for field-collected mussels (*Mytilus* spp.)

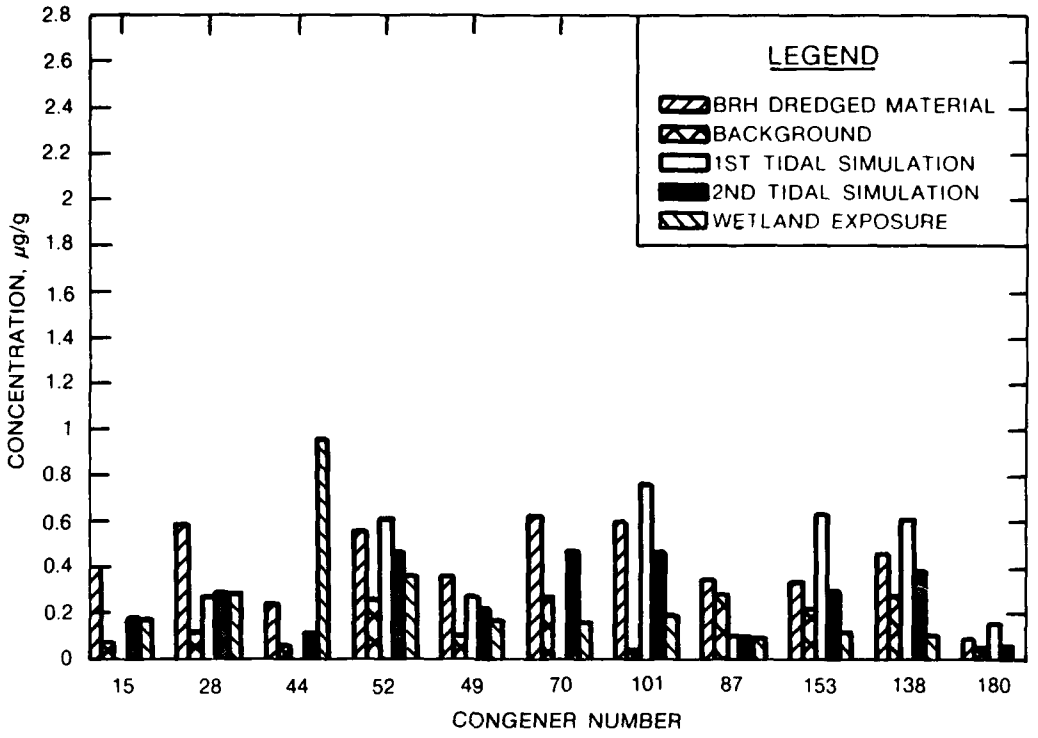


a. PCB congeners in mussels



b. PCB congeners in snails

Figure 13. Comparison of PCB congener concentrations in Black Rock dredged material, background tissue samples, animals exposed to BRH in two tidal simulation laboratory bioassays, and animals exposed to BRH in the wetland field site (Continued)



c. PCB congeners in sandworms

Figure 13. (Concluded)

(Goldberg et al. 1978) or clams (*Mercenaria mercenaria*) from New Bedford Harbor (Deubert, Rule, and Corte-Real 1981). Concentrations of PCBs in mussels collected in other field surveys ranged from an order of magnitude higher to two orders of magnitude lower than those reported in this study (Marchand, Vas, and Duursma 1976; Mower et al. 1977; Cowan 1981). Concentrations of PCBs in polychaetes from this study were within the range of values for polychaetes reported in previous laboratory studies with contaminated sediments (Fowler et al. 1978; Elder, Fowler, and Polikarpov 1979; McLeese, Metcalfe, and Pezzack 1980; Rubinstein, Lores, and Gregory 1983).

49. A comparison of the results of the initial static test, the tidal simulation bioassay tests, and the active biomonitoring data with that of the colonizing *N. succinea* is presented in Table 12. Field-collected sandworms colonizing the wetland showed copper and cadmium accumulation, which was predicted in the laboratory tests. The PCB and PAH bioaccumulation observed in laboratory bioassay tests was not observed in the field-collected sandworms as the tissue contents of PCBs and PAHs were below detection limits.

PART IV: CONCLUSIONS AND RECOMMENDATIONS

50. The estuarine marsh plant *S. alterniflora* was substituted for the freshwater index plant *Cyperus esculentus*, and the laboratory plant bioassay developed by Folsom and Lee (1981a, 1981b) for freshwater sediments was applied to an estuarine sediment. Uptake of selected heavy metals from the laboratory plant bioassay was compared to uptake by plants grown in the wetland field site. The laboratory plant data for copper and nickel accurately predicted levels of these metals in field-grown plants. The laboratory data accurately predicted zinc tissue levels in field-grown plants in 1985, but underpredicted zinc levels in 1986 field-grown plants. The laboratory plant data slightly underpredicted cadmium levels in field-grown plants. Laboratory-grown plants had lower chromium and lead concentrations than those measured in both the 1985 and 1986 field-grown plants. In general, the laboratory plant bioassay adequately predicted (i.e., field verified) heavy metal content of field-grown *S. alterniflora* during the first 2 years following dredged material disposal and marsh creation. It would be valuable to continue field verification of plant contaminant uptake for 5 years or until a stable physical and geochemical environment is reached in the wetland.

51. Copper, nickel, and chromium uptake by field-grown plants was accurately predicted by DTPA extraction data generated during the laboratory phase of the plant bioassay. Cadmium and zinc uptake by field-grown plants was greatly overpredicted by the laboratory DTPA extraction data. Decisions based on overprediction would be more environmentally protective than decisions based on underprediction. It is also of interest that heavy metal content of field-grown plants from the FVP wetland was not significantly higher than that of plants in natural marshes from surrounding areas or growing at the site prior to FVP wetland construction, with the possible exception of chromium. This indicates that for long-term impacts to plants, short-term assessments may be satisfactory.

52. The laboratory plant bioassay was shown to predict field levels of selected heavy metals accumulated by plants from one highly contaminated estuarine sediment during the first 2 years following disposal. The estuarine plant bioassay procedure should be evaluated using more sediments under varying salinity conditions, organic contents, metal contents, etc., to increase its applicability.

53. Static bioassay procedures were found to be inappropriate for predicting uptake of contaminants by animals exposed to Black Rock sediment to be used for creation of wetlands. Under static bioassay conditions, the sediment was toxic to the sandworms used in the test. Modification of the sediment by adding sand allowed the sandworms to survive, but the changes in the substrate brought about by the modification with sand raise questions about the validity and comparability of results to field data.

54. Bioassay procedures that utilized tidal exchange, similar to conditions that prevail at the wetland site, showed more promise than did static bioassays. Results of a preliminary tidal exchange bioassay showed acceptable animal survival with unmodified sediment and demonstrated uptake of PCBs and PAHs, especially by snails. In general, PAH concentrations were higher in snails than in sandworms exposed to Black Rock Harbor sediment in the tidal bioassay, although results could not be statistically analyzed due to lack of replication.

55. Following dredging and disposal activities, dredged material collected from the field site was used in a second tidal simulation laboratory bioassay. Use of site material in laboratory testing deviated from the underlying premise of the FVP, i.e., to make predictions derived from tests on sediment collected prior to dredging. However, prolonged development (due to additional testing) of laboratory wetland animal bioassay procedures and depletion of sediment collected for laboratory analysis precluded this option. Results of this second series of animal tidal bioassays suggested, as did the first tidal bioassay procedure, that there is a potential for wetland animals in contact with contaminated dredged material to accumulate contaminants.

56. Comparison of field-collected animal data with laboratory tidal bioassay data suggests that tidal simulation bioassay procedures are somewhat overpredictive of PCB congener, hexachlorobenzene, and DDE bioaccumulation in animals exposed to BRH dredged material in the wetland. No clear pattern between laboratory data and either of the field experiments emerged for metals.

57. Animal bioassay results form both static and tidal simulation tests indicate that tidal simulation procedures are superior to static tests for measuring contaminant uptake by organisms in the intertidal wetland habitat. However, wetlands consist of areas that are not regularly inundated by tidal action as well as areas that are not exposed to air during low tide, but

remain inundated. The existing tidal simulation test is a modification of existing static renewal bioassay procedures that provides for increased oxidation of the sediment surface. Comparison tests between aquatic flow-through bioassays and tidal bioassays should be conducted to determine if results of the two are comparable. If results are comparable, it would be cost effective to include wetland biota in aquatic flow-through bioassays conducted in predredging evaluations. This could significantly reduce the cost of required testing.

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Table 1

Summary of Animal Bioassay Test Procedures

Bioassay Procedure	Difficulties Encountered	Improvements
Initial static test	100-percent animal mortality	Modify procedure by adding sand
Modified static test	No tidal action, not real world	Add tidal action
Tidal I	System too large, no replication	Larger number of smaller systems
Tidal II	Limited replication, additional test development needed	Compare results of tidal simulation bioassay with flow-through aquatic bioassay (proposed problem remedy)

Table 2

Selected Physical and Chemical Parameters of Black Rock Harbor Sediment

Parameter	Value
Organic matter, %	19.5
Salinity, ppt	28.0
Electrical conductivity, dS/m	35.7
$CaCO_3$ equivalent, %	0.9
pH wet	7.6
pH reconstituted* air-dried	6.6
Lime requirement,** mg/g $CaCO_3$	4.8
Oil and grease, mg/g	17.5
Total sulfur, %	1.3

* Reconstituted air-dried pH is pH of a 1:2 sediment to solution suspension using air-dried sediment.

** Air-dried upland sediment limed to pH 7.0.

Table 3

Total Acid-Digestible and DTPA-Extractable Concentrations of Selected Metals in Black Rock Harbor Sediment

Heavy Metal	Concentration, µg/g: Total Acid-Digestible Wet Sediment (N = 3)		DTPA-Extractable: Wet Sediment (N = 4)			DTPA-Extractable: Air-Dried Sediment (N = 4)			ANOVA Results F	ANOVA Results P
Zn	1,370	(52.9)*	3.33	(0.509)	B**	962	(100)	A	3,715.04†	0.0001
Cd	23.3	(0.577)	0.047	(0.056)	B	28.7	(1.05)	A	2,954.34††	0.0001
Cu	2,860	(174)	0.473	(0.513)	B	387	(116)	A	95.07†	0.0001
Ni	203	(14.6)	7.59	(2.20)	B	66.9	(3.13)	A	961.44†	0.0001
Cr	1,403	(232)	0.313	(0.058)	B	0.828	(0.188)	A	48.83†	0.0004
Pb	399	(42.5)	0.175	(0.120)	B	16.3	(7.95)	A	160.17†	0.0001

* Mean (and standard deviation).

** Different letters in a row indicate significantly different means by ANOVA, $P < 0.05$.

† ANOVA performed on log 10-transformed data.

†† ANOVA performed on square root-transformed data.

Table 4

Leaf Tissue Content of Selected Heavy Metals in *S. alterniflora* Grown in the Laboratory and in the Field and Predicted from DTPA Sediment Extraction Data Using the Equations of Lee, Folsom, and Bates (1983)

Heavy Metal	Concentrations, µg/g												ANOVA	
	Laboratory (N = 4)			Field-Grown 1985 (N = 7)			Field-Grown 1986 (N = 7)			Predicted by DTPA (N = 3)			F	P
Zn	12.1*	(1.26)	C**	13.5	(5.03)	BC	19.2	(7.05)	B	41.7	(2.44)	A	15.89	0.0001
Cd	0.041	(0.007)	B	0.021	(0.046)	C	<0.0025	(0)	C	0.196	(0.026)	A	22.64	0.0001
Cu	4.02	(1.38)	A	5.65	(1.74)	A	7.48	(5.55)	A	2.70	(0.080)	A	2.20	0.1486
Ni	0.954	(0.388)	A	4.23	(6.13)	A	0.743	(0.675)	A	0.346	(0.0098)	A	2.26	0.1183
Cr	0.274	(0.322)	B	10.4	(8.21)	A	6.17	(5.49)	AB	1.63	(0.093)	AB	2.91	0.0648
Pb	0.237	(0.441)	B	3.45	(4.90)	A	0.945	(0.892)	AB	0.70	(0)	AB	3.18	0.0507

* Mean (and standard deviation).

** Letters in a row indicate statistical groupings of means (square root transformation, Waller-Duncan k-ratio t-test).

Table 5

Leaf Tissue Content of Selected Heavy Metals in *S. alterniflora* from Several Different Locations

Heavy Metal	Concentrations, µg/g												ANOVA Results	
	Natural Marsh* (N = 20)			Preconstruction (N = 7)			Field-Grown 1985 (N = 7)			Field-Grown 1986 (N = 7)			F	P
Zn	44.3**	(24.8)	A†	22.5	(9.53)	B	13.5	(5.03)	B	19.2	(7.05)	B	9.73	0.0001
Cd	0.203	(0.190)	A	0.173	(0.113)	A	0.021	(0.046)	B	<0.0025	(0)	B	12.33	0.0001
Cu	7.16	(2.16)	A	3.62	(1.18)	B	5.65	(1.74)	AB	7.48	(5.55)	AB	4.24	0.0114
Ni	2.47	(1.76)	B	5.64	(2.90)	A	4.23	(6.13)	AB	0.743	(0.675)	C	4.94	0.0055
Cr	3.41	(1.80)	A	1.11	(1.70)	B	10.4	(8.21)	A	6.17	(5.49)	A	5.02	0.0839
Pb	4.85	(6.47)	A	2.17	(0.834)	AB	3.45	(4.90)	AB	0.945	(0.892)	B	2.39	0.0839

* From Simmers et al. (1981).

** Mean (and standard deviation).

† Letters in a row indicate statistical groupings of means (square root transformation, Waller-Duncan k-ratio t-test).

Table 6

Concentration (Micrograms per Gram) of Selected Heavy Metals in Leaf Tissue of *Sporobolus virginicus* Grown in BRH Sediment Under Wetland Conditions

Heavy Metal	Laboratory	Field
Zn	26.2* (6.23)	No survival
Cd	0.857 (0.092)	
Cu	10.7 (5.62)	
Ni	6.82 (2.40)	
Cr	<0.025	
Pb	<0.013	

* Mean (and standard deviation) of four replicates.

Table 7

Comparison of Average Individual Weights (Grams, Wet Weight) of Sandworms (*Nereis virens*) at Day 0 and Day 14 from the Initial 14-Day Static Laboratory Bioassay

	Day 0 (N = 4)		Day 14 (N = 4)		ANOVA Results	
Substrate	Number of Worms	Average Weight per Worm	Number of Worms	Average Weight per Worm	F	P
Modified BRH	12.5	2.18* (0.30) A**	12.0	1.82 (0.32) A	2.78	0.146
BRR	11.5	2.38 (0.44) A	9.75	2.08 (0.69) A	0.53	0.492
Sand	10.3	2.78 (0.77) A	10.3	2.25 (0.73) A	0.99	0.359

* Mean (and standard deviation).

** Means in a row followed by the same letter indicate statistical groupings of means (ANOVA, $P < 0.05$).

Table 8

Initial 14-Day Static Laboratory Bioassay with the Sandworm (*Nereis virens*)*

Contaminant	Substrate: 25% BRH 75% Sand (N = 1)	Substrate: BRR (N = 1)	Substrate: Sand (N = 1)	14-Day Tissue Residues: 25% BRH 75% Sand (N = 4)	14-Day Tissue Residues: BRR (N = 4)	14-Day Tissue Residues: Sand (N = 4)	ANOVA Results F	ANOVA Results P
Heavy metals								
As	0.47	4.9	<0.49	1.08** (0.15) A	1.51 (1.15) A	1.02 (0.22) A	0.32†	0.7312
Cd	0.4	0.3	0.1	2.95 (0.26) A	1.82 (0.79) B	0.21 (0.04) C	129.66†	0.0001
Cr	92.3	230.5	0.9	10.33 (3.49) A	4.43 (2.40) B	6.43 (3.82) AB	3.33	0.0829
Cu	153.0	326.5	<4.9	153.8 (22.5) A	28.1 (6.22) B	17.6 (1.70) C	195.16†	0.0001
Ni	12.9	36.2	0.2	13.68 (3.08) A	6.23 (1.07) B	11.3 (4.18) AB	6.16	0.0206
Pb	22.4	75.2	13.3	5.68 (3.18) A	3.35 (1.10) A	5.48 (1.18) A	1.57	0.2605
Zn	83.6	257.5	<4.9	212.8 (43.5) A	244.8 (177.4) A	216.3 (81.7) A	0.09	0.9126
PCBs - Aroclor 1242	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0		
PAHs								
Phenanthrene	1.20	<0.32	<0.11	<0.45	<0.45	<2.8		
Anthracene	0.17	<0.32	<0.11	<0.45	<0.45	<2.8		
Fluoranthene	0.61	<0.32	<0.11	<0.45	<0.45	<2.8		
Pyrene	0.58	<0.32	<0.11	<0.45	<0.45	<2.8		
Benzo(a)-anthracene	0.44	<0.32	<0.11	<0.45	<0.45	<2.8		
Chrysene	0.38	<0.32	<0.11	<0.45	<0.45	<2.8		
Benzo(k)-fluoranthene	0.73	<0.32	<0.11	<0.45	<0.45	<2.8		
Benzo(a)pyrene	<0.38	<0.32	<0.11	<0.45	<0.45	<2.8		
Benzo(g,h,i)-perylene	<0.38	<0.80	<0.28	<1.0	<1.0	<1.0		
Ideno (1,2,3-c,d)-pyrene	<0.38	<0.80	<0.11	<1.0	<1.0	<1.0		

* Concentrations in micrograms per gram dry weight.

** Mean (and standard deviation). Letters in a row indicate statistical groupings of means (Waller-Duncan k-ratio t-test).

† ANOVA and means comparison procedures performed on log 10-transformed data.

Table 9

Concentrations of Selected Heavy Metals in Sandworms and Mudsnails Exposed to Contaminated Sediments in a Laboratory Tidal Simulation Chamber

Contaminant	Sand			Preconstruction FVP Site			Black Rock Reference			Black Rock Harbor		
	Substrate*	Sandworms**	Snails**	Substrate	Sandworms	Snails	Substrate	Sandworms	Snails	Substrate	Sandworms	Snails
Heavy metals												
Cd	0.21	0.841	8.59	0.73	0.874	11.2	3.6	0.822	13.4	22	1.02	18.2
Cu	0.19	16.5	2,913	43	36.0	3,762	380	20.4	4,823	2,590	16.7	5,920
Hg	0.14	0.420	0.255	0.24	0.335	0.219	0.81	0.363	0.240	3.5	0.206	0.264
As	0.16	21.3	13.7	2.8	23.7	18.1	8.4	22.5	19.9	12	18.8	24.6
PCBs												
2,4,4'-tri (28)†	BD††	BD	0.087	BD	BD	0.059	0.025	0.059	0.27	0.250	0.280	1.25
2,5,2',5'-tetra (52)			0.020		BD	0.087	0.027	0.101	0.27	0.340	0.620	1.29
2,4,2',5'-tetra (49)			0.004		BD	0.068	0.021	0.046	0.19	0.230	0.283	--
2,5,3',4'-tetra (70)			0.084		BD	0.014	0.036	BD	0.61	0.390	BD	2.61
2,4,5,2',5'-penta (101)			0.087		0.063	0.125	[illegible]143	0.140	0.37	0.430	0.770	1.55
2,3,4,2',5'-penta (87)			--		BD	--	0.026	BD	0.12	0.270	0.120	0.46
2,4,5,2',4',5'-hexa (153)		0.079	0.011		0.017	0.19	0.038	0.180	0.33	0.300	0.640	0.90
2,3,4,2',4',5'-hexa (138)		0.043	0.018		0.083	0.22	0.037	0.140	0.48	0.310	0.620	1.17
2,3,4,5,2',4',5'-hepta (180)		0.039	0.025		0.044	0.058	0.014	0.054	0.067	0.110	0.170	0.13
Total‡	--	≥0.161	0.336	--	≥0.297	0.821	0.267	≥0.720	2.507	2.630	≥3.503	9.36
PAHs												
Phenanthrene	0.050	0.039	0.090	0.10	0.058	0.15	0.48	0.055	0.14	6.9	0.11	0.67
Anthracene	0.0025	0.0018	0.0075	0.0099	0.0035	0.018	0.081	0.0027	0.026	2.3	0.016	0.25
Fluoranthene	BD	0.0089	0.034	0.10	0.044	0.35	0.077	0.055	0.84	8.1	0.14	4.2
Pyrene		BD	BD	0.094	0.031	0.24	0.92	0.058	BD	9.2	0.18	6.7
3,6-dimethylphenanthrene			BD	BD	BD	0.083	BD	0.030	1.2	0.020	0.19	
Triphenylene			0.033	BD	0.024	0.20	BD	0.030	BD	BD	0.18	BD
Benzo(b)fluorene			BD	0.017	BD	BD	0.088	BD	0.033	1.4	0.017	0.52
Benzo(a)anthracene			0.019	0.072	0.0070	BD	0.52	0.0063	0.25	6.3	0.080	0.95
Chrysene			BD	0.067	BD	0.072	0.50	BD	0.12	5.5	0.085	1.7
Benzo(e)pyrene			BD	0.053	BD	BD	0.40	BD	BD	3.2	BD	0.26
Benzo(j)fluoranthene			BD	BD	BD	BD	BD	BD	BD	BD	BD	BD
Perylene			BD	0.016	0.0017	0.0052	0.16	BD	BD	1.1	0.0026	0.060
Benzo(b)fluoranthene			BD	0.068	0.010	0.032	0.69	0.0072	BD	4.0	0.12	0.81
Benzo(k)fluoranthene		0.0035	0.0067	0.046	0.0096	0.029	0.36	0.0098	0.13	2.2	0.15	0.55
Benzo(a)pyrene		BD	BD	0.067	0.0087	0.013	0.88	0.0063	0.077	4.9	0.23	0.98
Dibenzo(a,j)anthracene		BD	BD	0.041	BD	BD	BD	BD	BD	1.4	BD	0.29
Dibenzo(a,i)pyrene		BD	BD	BD	BD	BD	0.21	BD	BD	1.6	BD	BD
Benzo(g,h,i)perylene	0.82	0.21	0.24	BD	0.17	0.27	1.5	0.17	0.34	5.5	0.17	0.47
Indeno(1,2,3-c,d)pyrene	BD	BD	BD	0.045	BD	BD	0.63	BD	BD	4.3	BD	0.31
3-methylcholanthrene	BD	BD	BD	BD	BD	BD	BD	BD	BD	BD	BD	0.034
Anthanthrene	BD	BD	BD	0.014	BD	BD	0.13	BD	BD	0.92	BD	0.021

* Substrate concentrations are expressed in micrograms per gram oven-dry weight.
** Animal concentrations are expressed in micrograms per gram oven-dry weight.
† International Union of Pure and Applied Chemists (IUPAC) number.
†† Below detection limits.
‡ Total of the nine congeners analyzed.

Table 10

Contaminants in BRH Dredged Material and Contaminant Uptake by Animals Grown in BRH Dredged Material from the Wetland Under Laboratory Tidal Simulation Conditions

Contaminant	Black Rock Dredged Material* (N = 2)	Mussels** Background (N = 1)	Mussels** Sand Control (N = 2)	Mussels** Black Rock (N = 2)	Snails Background (N = 1)	Snails Sand Control (N = 2)	Snails Black Rock (N = 2)	Sandworms Background (N = 1)	Sandworms Sand Control (N = 2)	Sandworms Black Rock (N = 2)
Heavy metals										
Cd	20.84† (0.01)	15.8	26.0 (5.4)	22.5 (2.7)	14.1	1.09 (1.9)	9.7 (3.0)	0.65	1.4 (1.0)	0.713 (0.015)
Cr	1,746 (31)	0.64	0.91 (0.03)	16.4 (17.9)	15.3	1.4 (0.8)	7.6 (6.0)	0.32	0.34 (0.08)	2.82 (0.41)
Cu	2,524 (66)	14.2	55.3 (9.1)	91.6 (48.7)	822	3,035 (630)	3,006 (1153)	7.9	19.6 (2.0)	69.5 (0.2)
PCBs										
4,4'-dichlorobiphenyl (15)††	0.409 (0.028)	0.149	0.129 (0.025)	0.433 (0.063)	0.220	0.066 (0.003)	0.257 (0.026)	0.081	<0.008	0.089 (0.002)
2,4,4'-trichlorobiphenyl (28)	0.593 (0.045)	0.150	0.151 (0.017)	0.486 (0.063)	0.258	0.072 (0.014)	0.624 (0.076)	0.126	0.030 (0.009)	0.092 (0.014)
2,3,2',5'-tetrachlorobiphenyl (44)	0.254 (0.033)	0.050	0.097 (0.032)	0.103 (0.026)	0.073	0.042 (0.014)	0.104 (0.015)	0.070	<0.036	0.023 (0.002)
2,5,2',5'-tetrachlorobiphenyl (52)	0.596 (0.052)	0.233	0.220 (0.015)	0.632 (0.081)	0.309	0.048 (0.004)	0.571 (0.046)	0.266	0.027 (0.006)	0.478 (0.013)
2,4,2',5'-tetrachlorobiphenyl (49)	0.374 (0.028)	0.162	0.148 (0.011)	0.391 (0.056)	0.172	0.033 (0.005)	0.334 (0.026)	0.108	0.016 (0.005)	0.237 (0.005)
2,5,3',4'-tetrachlorobiphenyl (70)	0.637 (0.054)	0.225	0.209 (0.011)	0.662 (0.127)	0.405	0.040 (0.005)	0.707 (0.094)	0.283	0.024 (0.002)	0.482 (0.006)
2,4,5,2',5'-pentachlorobiphenyl (101)	0.607 (0.058)	0.221	0.211 (0.007)	0.604 (0.090)	0.427	0.039 (0.003)	0.655 (0.069)	0.290	0.016 (0.004)	0.476 (0.014)
2,3,4,2',5'-pentachlorobiphenyl (87)	0.355 (0.035)	0.075	0.062 (0.003)	0.297 (0.060)	0.060	<0.007	0.194 (0.040)	0.047	<0.008	0.109 (0.003)
2,4,5,2',4',5'-hexachlorobiphenyl (153)	0.346 (0.048)	0.171	0.170 (0.002)	0.625 (0.457)	0.325	0.049 (0.000)	0.379	0.226	0.045 (0.006)	0.307 (0.001)
2,3,4,2',4',5'-hexachlorobiphenyl (138)	0.471 (0.060)	0.143	0.136 (0.006)	0.366 (0.056)	0.325	0.043 (0.000)	0.460 (0.059)	0.282	0.033 (0.007)	0.393 (0.003)
2,3,4,5,2',4',5'-heptachlorobiphenyl (180)	0.101 (0.019)	0.021	0.015 (0.004)	0.035 (0.009)	0.069	0.017 (0.003)	0.091 (0.010)	0.059	0.023 (0.002)	0.075 (0.002)
Total‡	4.70 (0.48)	1.6	1.5 (0.3)	4.6 (1.1)	2.6	<0.46 (0.01)	4.3 (0.5)	1.8	<0.27 (0.08)	3.2 (0.045)
Hexachlorobenzene	0.77 (0.035)	0.062	0.070 (0.025)	0.051 (0.020)	0.064	0.018 (0.015)	0.025 (0.012)	0.017	0.025 (0.016)	0.026 (0.002)
o,p'-DDE	0.315 (0.004)	0.116	0.107 (0.007)	0.591 (0.401)	0.251	0.023 (0.000)	0.431 (0.052)	0.165	<0.010 (0.002)	0.282 (0.007)
p,p'-DDE	0.232 (0.021)	0.079	0.081 (0.005)	0.245 (0.031)	0.122	0.012 (0.001)	0.239 (0.039)	0.069	<0.008 (0.001)	0.117 (0.005)
PAHs										
Phenanthrene	3.04 (0.03)	2.61	5.04 (1.83)	4.05 (1.83)	0.61	0.737 (0.303)	0.36 (0.09)	0.15	0.735 (0.699)	0.22 (0.024)
Anthracene	0.637 (0.037)	0.057	0.049 (0.025)	0.279 (0.076)	0.055	0.024 (0.009)	0.141 (0.025)	0.005	0.026 (0.028)	0.021 (0.001)
Fluoranthene	5.82 (0.10)	1.4	0.541 (0.162)	3.3 (0.2)	0.177	0.087 (0.001)	2.2 (0.7)	0.051	0.126 (0.135)	0.306 (0.044)
Pyrene	5.10 (0.61)	1.12	0.657 (0.130)	4.7 (0.7)	0.27	<0.18	3.1 (0.7)	0.039	<0.121	0.35 (0.057)
3,6-dimethylphenanthrene	0.265 (0.098)	0.484	0.503 (0.083)	0.785 (0.155)	<0.044	<0.029	0.152 (0.060)	<0.010	<0.014	<0.015
Triphenylene	<0.098	0.168	<0.048	1.34 (0.05)	<0.130	<0.089	0.611 (0.133)	0.034	<0.044	0.235 (0.036)
Benzo(b)fluorene	0.370 (0.086)	0.149	0.085 (0.012)	0.528 (0.037)	<0.058	<0.038	0.127 (0.016)	<0.012	<0.018	<0.021
Benzo(a)anthracene	3.34 (0.35)	0.605	0.294 (0.047)	1.99 (0.03)	0.091	<0.037	0.478 (0.155)	0.016	<0.017	0.061 (0.001)
Chrysene	2.42 (0.12)	0.382	0.131 (0.026)	1.59 (0.17)	0.084	<0.048	0.533 (0.165)	<0.016	<0.024	0.070 (0.003)
Benzo(e)pyrene	1.58 (0.05)	0.559	0.329 (0.015)	1.14 (0.05)	<0.233	<0.152	0.344 (0.101)	<0.051	<0.076	<0.099
Benzo(j)fluoranthene	<0.817	<0.419	<0.372	<0.434	<1.023	<0.688	<0.343	<0.207	<1.130	<0.345
Perylene	0.709 (0.013)	0.054	0.025 (0.007)	0.124 (0.005)	<0.012	<0.008	0.020 (0.001)	<0.003	<0.004	<0.004
Benzo(b)fluoranthene	2.56 (0.08)	0.251	0.084 (0.027)	0.776 (0.057)	<0.023	<0.015	0.312 (0.057)	0.005	<0.008	0.025 (0.005)
Benzo(k)fluoranthene	1.33 (0.10)	0.071	0.021 (0.008)	0.279 (0.005)	0.028	0.024	0.213 (0.041)	0.012	0.011 (0.003)	0.020 (0.001)
Benzo(a)pyrene	2.24 (0.14)	0.036	0.010 (0.002)	0.279 (0.019)	<0.014	<0.009	0.210 (0.036)	<0.003	<0.005	0.014 (0.001)
Dibenzo(a,j)anthracene	<0.919	<0.043	<0.036	<0.043	<0.102	<0.066	<0.034	<0.021	<0.033	<0.039
Dibenzo(a,i)pyrene	0.386 (0.444)	0.158	<0.144	0.483 (0.059)	<0.326	<0.219	0.282 (0.071)	<0.071	<0.105	<0.111
Benzo(g,h,i)perylene	3.90 (0.74)	0.289	0.20‡‡	0.632 (0.032)	<0.149	<0.103	0.261 (0.032)	0.084	<0.054	<0.084
Indeno(1,2,3-c,d)pyrene	2.80 (0.10)	<0.041	<0.036	0.079 (0.003)	<0.102	<0.066	0.155 (0.036)	<0.021	<0.032	<0.034
3-methylcholanthrene	0.016 (0.003)	<0.021	<0.018	<0.021	<0.050	<0.034	<0.030	<0.010	<0.016	<0.018
Perylene	0.589 (0.000)	<0.008	<0.007	0.014 (0.001)	<0.020	<0.013	0.011 (0.001)	<0.004	<0.006	<0.007

* Concentration is expressed in micrograms per gram oven-dry weight.
** Concentration is expressed in micrograms per gram ash-free dry weight.
† Mean (and standard deviation).
†† IUPAC number.
‡ Total of the 11 congeners analyzed.
‡‡ Only one value available.

Table 11

Contaminant Concentrations in Animals Exposed for 28 Days to BRH Dredged Material at the Field Wetland Site

Contaminant		Black Rock Dredged Material*	Mussels		Snails		Sandworms	
			Background**	Field**	Background**	Field**	Background**	Field**
4,4-dichlorobiphenyl	(15)†	0.408	0.149	0.307 (0.048)††	0.220	0.070 (0.028)†	0.081	0.182 (0.010)
2,4,4'-trichlorobiphenyl	(28)	0.593	0.150	0.301 (0.038)	0.258	0.022 (0.008)	0.126	0.295 (0.011)
2,3,2',5'-tetrachlorobiphenyl	(44)	0.254	0.050	0.681 (0.105)	0.073	0.444 (0.033)	0.070	0.959 (0.001)
2,5,2',5'-tetrachlorobiphenyl	(52)	0.568	0.233	0.462 (0.034)	0.309	0.250 (0.008)	0.266	0.372 (0.019)
2,4,2',5'-tetrachlorobiphenyl	(49)	0.373	0.162	0.305 (0.016)	0.172	0.109 (0.012)	0.108	0.176 (0.012)
2,5,3',4'-tetrachlorobiphenyl	(70)	0.637	0.225	0.461 (0.071)	0.405	0.271 (0.005)	0.283	0.173 (0.027)
2,4,5,2',5'-pentachlorobiphenyl	(101)	0.607	0.075	0.295 (0.053)	0.060	0.175 (0.003)	0.047	0.201 (0.010)
2,3,4,2',5'-pentachlorobiphenyl	(87)	0.355	0.221	0.195 (0.032)	0.427	0.069 (0.010)	0.290	0.103 (0.022)
2,4,5,2',4',5'-hexachlorobiphenyl	(153)	0.346	0.171	0.277 (0.014)	0.325	0.188 (0.016)	0.226	0.126 (0.005)
2,3,4,2',4',5'-hexachlorobiphenyl	(138)	0.470	0.143	0.242 (0.020)	0.325	0.186 (0.026)	0.282	0.118 (0.007)
2,3,4,5,2',4',5'-hexachlorobiphenyl	(180)	0.100	0.021	0.029 (0.002)	0.069	0.058 (0.000)	0.059	0.028 (0.001)
Total‡		4.7	1.6	3.55 (0.43)	2.6	2.04 (0.05)	1.8	2.73 (0.12)
Hexachlorobenzene		0.077	0.062	0.010 (0.001)	0.064	0.007 (0.001)	0.017	0.014 (0.001)
o,p'-DDE		0.315	0.116	0.153 (0.018)	0.251	0.097 (0.006)	0.165	0.087 (0.005)
p,p'-DDE		0.231	0.079	0.130 (0.016)	0.122	0.086 (0.013)	0.069	0.047 (0.008)

* Concentrations are expressed in micrograms per gram oven-dry weight.
** Concentrations are expressed in micrograms per gram ash-free dry weight.
† IUPAC number.
†† Mean (and standard deviation) of two replicates.
‡ Total of the 11 congeners analyzed.

APPENDIX A: POSTCONSTRUCTION FIELD SITE STUDIES

General Characteristics of the Field Site

1. Within approximately 9 months after the dredging operations, the dredged material in the wetland had stabilized at the desired surface elevation, and by fall 1986 the wetland field site had developed into a *Spartina alterniflora* marsh/mudflat typical of the Long Island Sound area. Over the period 1983-87, the planted *S. alterniflora* became well established, and small tidal creeks developed. Currently, the tidal creek moves in through the weir and inundates the site via a single tidal creek and its branches. The dredged material as of spring 1987 was not appreciably consolidated. The material has remained quite fluid, as has been observed in other dredged material marsh creations that have employed uncontaminated material. The consolidation is sufficient, however, to support intertidal invertebrates, the tubes of burrowing invertebrates, and the weight of wading birds--but not that of large mammals or humans.

2. At normal high tide the site is almost completely flooded. Only part of the mounded coarse-grained material near the outflow of the dredge pipe remains above the water. The tidal inundation generally floods all the *S. alterniflora*-covered areas of the site. At normal low tide the field site is completely drained, and the entire surface is exposed to the atmosphere. There is a 1.5- to 2-hr time lag between tidal flow outside the Tongue Point perimeter dike and the field site, both during filling and draining. Consequently, the tidal inundation is approximately the same depth as that observed on the preconstruction wetland at the same location.

3. In general, the surface conditions of the site appear to be very similar to those at the nearby salt marsh/mudflat ecosystems used as references.

Plant Establishment and Colonization

4. The planting of the *S. alterniflora* with the attached background marsh material on the east side of the bridge was only marginally successful. The planting materials were too heavy, and many sank into the unconsolidated dredged material. Although all the plant materials that remained on the

surface did grow, the resulting plant cover was somewhat sparse. The *S. alterniflora* planted on the west side of the bridge was supplied in lightweight peat pots. The plants in peat pots did not sink as easily, and therefore a denser stand resulted than that on the east side of the bridge.

5. In both planting areas, the plants would probably have benefited from a spring planting rather than the winter planting that was used. Certainly plant survival would have been enhanced if the dredged material had been allowed to consolidate to a greater extent or if some material had been used to physically support the propagules in the fluid material.

6. During 1985, *Salicornia* (glasswort) was observed to be colonizing the wetland. This was anticipated, since *Salicornia* is present in approximately the same ecological niche in the remainder of the Tongue Point endiked area. In 1987, the *Salicornia* was still present on the wetland, and the *S. alterniflora* stand was still vigorous and expanding in the higher portions of the endiked wetland creation. It is anticipated that the *S. alterniflora* will eventually cover the entire area with the exception of the tidal creek and its predominant tributaries.

Animal Colonization

7. Animal colonization appeared to be limited to tide-borne organisms. A large population of any single resident species was not observed prior to 1985. During that year, it was noted that a large population of the sandworm, *Nereis succinea*, had become established. *Nereis succinea* is similar to *Nereis virens*, the WES bioassay organism, and is indigenous to the mudflats of the central Atlantic coast. It is a burrowing animal usually present in small numbers in the Long Island Sound area. It is important to note that by 1985 the *N. succinea* population in the field site had become quite dense. In 1986 the *N. succinea* population was still dense, and numerous individuals in reproductive status were observed. There is apparently a very large and actively reproducing population at the FVP wetland.

8. It was expected that the mud snail (*Nassarius obsoletus*) and the mussel (*Modiolus*) would colonize the site, as they are prevalent species in the area within the Tongue Point endikement. However, none were present at the time this report was prepared. There is a possibility that the sandworm (*N. succinea*), which was not a prevalent species prior to construction, has

been favored by the physical consistency of the dredged material or is better equipped to colonize via the weir, or both. The lack of either snails or mussels may be related either to the dike (as a barrier) or to unsuitable conditions (temperature, oxygen, contaminants, etc.) within the site for survival of larval or juvenile molluscs.

9. Following the creation of the wetland, other aquatic animals were seen moving through the site with the tidal waters. Among these were fishes and horseshoe crabs. Apparently, colonization and site usage by many of the local aquatic and salt marsh invertebrates depend upon the ability of those species to float or swim through the weir with the tide.

10. A large number of more mobile species have been observed using the field site. Most of these were birds, for whom the added wetland habitat has apparently served as a food source (Table A1). Some mammals have also been observed. In general, there does not appear to be a great alteration of the animal usage of the field site following the wetland creation.

Table A1

Vertebrate Usage of the FVP Wetland Field Site

(Observed, August 1984)

Bird Species
Nycticorax nycticorax--Black crowned night heron
Falco sparvevius--American kestrel
Parus carolinensis--Carolina chickadee
Melospiza melodia--Song sparrow
Passerculus sandwichensis--Savannah sparrow
Passer domesticus--House sparrow (weaver finch)
Sturnus vulgaris-- Starling
Cyanocitta cristata--Bluejay
Mimus polyglottos--Mockingbird
Corvus brachyrhynchos--Common crow
Corvus ossifragus--Fish crow
Zenaida macroura--Mourning dove
Ceryle alcyon--Belted kingfisher
Larus argentatus--Herring gull
Larus delawarensis--Ringbilled gull
Anas rubripes--Black duck
Nyctea scandiaca--Snowy owl
Mammal Species
Rattus norvegicus--Norway rat
Peromyscus maniculatus--White footed deermouse
Neotoma sp.--Wood rat